Ecological and Beneficial Use Assessment of Farmington Bay Wetlands:

Assessment and Site-Specific Nutrient Criteria Methods Development

Phase I

Progress Report to EPA, Region VIII
and
Final Report for Grant:
CD988706-03

Submitted by

Theron G. Miller, Ph.D.
Utah DEQ, Division of Water Quality
and
Heidi M. Hoven, Ph.D.
The Institute for Watershed Sciences

April 10, 2007

Table of Contents

List of Figures

List of Tables

Table **Page**

Executive Summary

Background and Purpose of Study

There has been a growing concern by waterfowl managers, scientists and citizen groups that the nutrient load from wastewater discharges along the Wasatch front may be exceeding the assimilatory capacity of the wetland and Farmington Bay ecosystems. Concurrent with this growing concern, EPA has been encouraging states to develop methods for assessing wetland condition. Utah DWQ applied for and received three Wetlands Protection Grants starting with the 2004 field season. The primary objective of this study is to develop assessment methods that will be used to perform §305(b)/303(d) assessments. This process will include establishing site-specific criteria for phosphorus. Total nitrogen never exceeded Utah's narrative standard of 5 mg L^{-1} and it was indeed often below instrument detection limits of 0.05 mg L^{-1}. Therefore, although the wetlands and Farmington Bay were nearly always nitrogen limited, it is unlikely that treatment options would reduce ambient nitrogen concentrations and therefore change ecological conditions of the Bay or wetlands. Additional evidence for this conclusion is the frequent dominance of nitrogen-fixing Cyanobacteria throughout the Bay and wetlands, which would negate any nitrogen removal in wastewater treatment systems.

This effort represents one of the first attempts by any of the states to establish water quality standards and methods for wetlands 303(d) assessment. This is primarily because wetlands assessment methods development is in its infancy and there is a dearth of data describing the relationship between nutrient gradients and biological responses in wetlands, particularly Great Salt Lake wetlands. Therefore, our goals are twofold: 1) test existing parameters outlined in EPA's various assessment modules and other potentially useful parameters for their utility in assessing Great Salt Lake wetlands; and 2) Develop metrics and ultimately an Index of Biological Integrity (IBI) that will identify thresholds of significant change (impairment) that can be attributed to nutrients. These thresholds will then be used to set a site-specific water quality standard for phosphorus and simultaneously used to determine beneficial use support status.

The initial wetlands study design focused on measuring nutrient attenuation along a longitudinal gradient established by water passing through successive impoundments or at increasing distances across the mudflats from POTW discharges. We identified reference (least impacted) as well as target (nutrient enriched) sampling sites. Particular biotic parameters that we focused on include: macrophytes (percent cover, stem height, species composition, tissue nutrient concentrations and ratios, above ground biomass) phytoplankton and periphyton community structure; macroinvertebrate community composition and shorebird nesting success and forage preference studies. Abiotic factors in the water include total phosphorus (P), nitrate-nitrite (N), ammonia, metal concentrations, pH, electrical conductance (EC), dissolved oxygen (DO) and temperature. Sediment nutrient concentrations, organic carbon, pH and EC were also measured.

Several reports were prepared by individual contractors (see Appendices). This report summarizes and assimilates the wetland reports and data and includes additional analyses pertaining to wetland function and nutrient dynamics. Potential metrics are reviewed and additional data gaps are identified that will increase the accuracy of the wetland assessment.

Similar reports that assimilate the existing data and analyses for the open water and for selenium speciation and partitioning in the wetlands will be released in the coming few months.

Plant Community Responses to Water Quality at Impounded and Sheetflow Sites

The two wetland types that we studied are impounded and sheetflow wetlands. Among our impounded wetland sites, pH rarely exceeded 9 and some of these measurements were made in Public Shooting Grounds (reference) ponds that had less than 0.05 mg L^{-1} total P. Yet, DO in the Public Shooting Grounds was often 120% to 200% saturation. In this case, the source of high dissolved oxygen was the dense meadows of the rooted and submerged vascular plant *Stuckenia* sp. (sego or fineleaf pondweed), and a calcareous green macroalga, *Chara* sp. Low nutrient and high dissolved oxygen concentrations with dense vegetative cover was a common condition among the impounded wetlands at our reference site.

Plant communities in the targeted impounded sites experienced important differences when compared to the reference ponds. Specifically, submerged aquatic vegetation (SAV), primarily *Stuckenia,* demonstrated a premature senescence during August. This amounted to more than 50% loss in aerial cover. Notably, this was before the arrival of waterfowl migrants. Extensive surface mats of filamentous algae or duckweed often developed on these ponds and heavy coatings of biofilms (composed of epiphytic algae, sediment, and possibly bacteria and fungi) were observed on the living leaves. This surface and epiphytic shading may reduce light penetration to below optimal or even threshold requirements. Further, this would expectedly be exacerbated by shorter photoperiod and lower sun angle as fall progressed. If photosynthesis rates are sub-optimal (i.e. $P<R$), there may not be adequate oxygen production to diffuse down to the roots and maintain an oxygen-rich root zone. There was also a concomitant reduction in macroinvertebrate species (primarily odonates and amphipods) that typically inhabit these underwater meadows. In turn, this could represent a decline in additional food availability during a time when waterfowl are attempting to nourish and regain energy stores. These observations warrant further investigation, particularly as to the seasonal timing and whether correlations between nutrient (water column and SAV tissues), light attenuation, and biomass (as Chlorophyll a (Chl a) of phytoplankton and epiphytes, and g dry weight SAV per unit area) exist. If such correlations are documented, they need to be quantified and considered for inclusion into an IBI. In addition, recent literature indicates that Photosystem II fluorescence is a useful indicator of stress (shading, etc.), and exhibits potential as an SAV community metrics of wetland condition. Ultimately, our concern is that large underwater meadows of *Stuckenia* (a preferred food by waterfowl) and associated macroinvertebrates may be largely disappearing prior to the arrival of migrating of waterfowl.

The sheetflow target sites (Publicly Owned treatment Works (POTW) discharges) are very different from the impoundments in their structure and characteristics. These sites contained relatively low pH values (circa 7.4-7.7), while total P ranged from 2 and 4 mg L^{-1} along the transects. Such high nutrient loads would be expected to cause very high levels of primary production that would also be associated with large diel swings in DO and pH, and noxious plant or algal blooms. Although DO at these sites fell to near 2 mg L^{-1} during evening hours, daytime values never exceeded saturation. In addition, although luxuriant stands of Phragmites and cattails occurred at the sheet flow target sites, there was very little attenuation of phosphorus along the range of sampling sites. This apparent lack of uptake from the water column in the midst of luxuriant emergent vegetation supports the paradigm that sediment is the primary source of nutrients for emergent macrophytes and secondly, that either sediment or plant binding/uptake sites are saturated and only a small amount of assimilation by the system is occurring. Both Phragmites and cattail (the dominant species at target sites) are well documented for removing nutrient burdens from water as a form of treatment and are acceptable in performing such functions. However, current P loading rates (~ 8-12 g m^{-2}) exceed recommended values (2 -4 g m^{-2}). This brings into question the actual efficacy of Phragmites and cattail to remove nutrients in this situation. Further assessment of biomass relative to nutrient loading and assimilative capacity of nutrients by emergent species would provide a metric for determining whether sheetflow sites outside of POTW and State Wildlife Management Areas (WMAs) are capable of treating nutrient enriched water.

In addition, one of the major metrics suggested in EPA's "*Methods for Evaluating Wetland Condition*" modules is changes in species composition to invasive/exotic species and a reduction in species richness. Although Phragmites/Typha communities occur adjacent to the POTW discharges they are not the dominant vegetation type at "downstream" sites. Rather, the succession we have observed since lake levels have subsided (2002-2006) is colonization by the salt-tolerant Salicornia and alkali bulrush (two native non-invasive species). Hence, the question remains: Will we eventually see dominance of these sheetflow sites by the more aggressive Phragmites and Typha as sediment salts continue to be flushed by fresh water? This important question cannot be answered in the short time-frame of this study and hence, this important metric is unavailable for inclusion in our IBI.

Finally, although Cyanobacteria were common among sampling sites, there were also diverse populations of diatoms, an algal group typically identified with mesotrophic or oligotrophic streams and lakes. Diatoms are well known indicators of water quality but identifying key species or assemblages of species takes seasonal sampling for several years. Additional literature and field research will be performed to determine whether the species found in Farmington Bay wetlands respond to the different nutrient concentrations in a predictive manner. This information will enhance our interpretation of ecological processes and provide more accurate assessments of water quality and beneficial use support for these wetland habitats.

Macroinvertebrate Response to Water Quality

Tolerance values for individual taxa and statistical analysis revealed important information. For example, mayflies (Ephemeroptera; the most sensitive taxa among all of the samples), were quite tolerant of Electrical conductance (EC) values up to10,000 umhos cm^{-2} and pH values to about 9.5. Yet, mayfly numbers were depressed or absent from enriched (high total P and low DO) sites. Mayfly relative abundance provides one of the most sensitive metrics of this study. Dragonflies and damselflies (Odonata) were sensitive to high salinity but, as a group, were tolerant of a broader range of other physical/chemical variables than mayflies. As expected, midges (Chironomidae) and water boatman (Corixidae) were the most ubiquitous and numerous taxa among all of the study sites. Their known wide tolerance range to various physical and chemical parameters would explain this occurrence. The only locations where midges were noticeably absent were the impoundments of the FB WMA and in the first three sites leading from the Central Davis Sewer District (CDSD). Water boatmen were similarly absent from the CDSD sites but were abundant at all other sites. When these sample data were aligned with Davis County mosquito spray records, we noted frequent spray events of both the larvacide BTI, as well as broad-spectrum adulticides at these locations. Careful co-located sampling of the pesticides and macroinvertebrates following spray events will be performed during 2007 to assess the importance of spraying. Despite large variances among samples, macroinvertebrate community responses appear to be useful metrics in developing an Index of Biological Integrity. These include: Relative abundance of Ephemeroptera (mayflies), and Odonata, relative abundance of the collector-gatherer feeding, and predator feeding groups, percent clingers and others. This will be enhanced by additional environmental tolerance information that is expected to be released shortly by EPA.

Shorebird Nesting Success and Prey Selection

A very intensive study of nesting and hatching success and prey availability and selection for black-necked stilts and American Avocet was performed. Approximately 3500 nests were marked and monitored. More that 95% of the nests located in Farmington Bay successfully produced offspring. Similarly, 96% of the eggs that survived until time-of-hatching, successfully hatched. These values were similar to those found in Bear River National Migratory Bird Refuge and which are the highest success rates ever recorded in the nation. These high success rates, however, are partly attributed to the aggressive depredation program operated by the US Fish and Wildlife Service and Utah Division of Wildlife Resources. Comparison of stomach analysis with ambient macroinvertebrate sampling indicated that corixids and midges were the most common prey item. As mentioned above, these were the two most common taxa throughout all of our study sites.

Nutrient Dynamics and Sediment Phosphorus Studies

There were two major observations concerning nutrient measurements: 1) Except for stations very near the POTW discharges, all sampling sites exhibited severe nitrogen limitation. (i.e. P was in great abundance for plant growth relative to N and C). Indeed, in many of the impoundments, N was undetectable (<0.05 mg L^{-1}); and 2) Contrary to our hypothesis, a substantial gradient only occurred in the four successive ponds at the Ambassador Duck Club complex and to a lesser extent, in the Newstate Duck Club complex. Attenuation in these ponds is attributed to relatively much longer retention times than the other impoundment systems. Throughout the other locations, there was only a slight decrease in water column P. Among the sheetflow sites, there was also rapid attenuation in N and again, only a slight reduction in P. Phosphorus concentrations remained within about 20% of those found in the upstream locations. This lack of nutrient attenuation is attributed to the exceedence of nutrient uptake potential by wetland vegetation and to saturation of binding sites in the sediments. Sediment samples contained 300 to 1200 mg kg^{-1} total P. Further, biologically available (soluble) P ranged from 10 to 80 mg kg^{-1} in the sediments- indicating that there is continual exchange with the water column.

Preliminary Conclusions

Results of the sheetflow plant and macroinvertebrate studies, the shorebird forage preference studies and the notable nesting and hatching success, suggest that the Farmington Bay sheetflow wetlands are supporting the beneficial use of support for waterfowl and shorebirds and the aquatic life in their foodchain. In addition, the large flocks of numerous species of shorebirds that congregate and aggressively feed during migratory staging adds further support for this conclusion. One additional set of information that would complete this study of the life history stages of shorebirds in Farmington Bay wetlands would be a detailed characterization of juvenile habitat and forage preference. This important period of time has received little attention in Great Salt Lake wetlands or other shorebird nesting colonies elsewhere in the US. Secondly, additional water/sediment/plant nutrient studies need to be performed to better understand nutrient movement throughout the sheetflow sites. This study will include an assessment of the potential for continued conversion from alkali bulrush/Salicornia community to a Phragmites/cattail community.

Results of the impounded wetland studies remain inconclusive. The extensive epiphyte cover on SAV leaves, surface mats of duckweed and filamentous algae, and the premature SAV senescence in the upstream ponds appear to be related to the high nutrient concentrations experienced at the target sites. This relationship needs to be verified and careful observations need to be made ascertaining the timing of senescence with arrival and residence time of migrating waterfowl.

Potential metrics for wetlands assessment

Based upon the data analysis thus far, candidate parameters for a multimetric index of biological integrity, as well as provide the essential data set for establishing an appropriate site-specific water quality standard for phosphorus include:

1. Macroinvertebrate species composition and density (during nesting season and fall migration season).
2. Percent of Ephemeroptera
3. Percent of Chironomidae
4. Percent Odonates or clingers
5. Percent exotic and/or invasive plants
6. Submerged aquatic vegetation above ground biomass
7. SAV percent coverage
8. C:N:P ratios in phytoplankton and macrophytes
9. Chlorophyll a / macrophyte fluorescence
10. turbidity/ light penetration
11. Presence/composition of floating vegetation
12. Summer mean diel DO
13. Diel minimum DO
14. Water column and sediment H_2S measurements

These parameters include most of those recommended by EPA (2002) and several that appear to be uniquely responsive in GSL wetlands. These will be measured during the 2007 field season.

In addition, data assemblages from 2004, 2005, 2006 and 2007 will be tested separately and in total to evaluate the reproducibility and representativeness of the data set.

Finally, Reports by Rushforth (Appendix D) and Wurtzbaugh (Appendix E) are also appended to this report to display the additional research that has been performed on Farmington Bay wetlands and open-water environments. However, detailed analysis and interpretation, such as presented here, are not included in this report. Rather, additional sample collection, data analysis and reporting will be provided by the end of 2007.

1.0 Introduction

The Great Salt Lake and its associated wetlands serve as breeding habitat and migratory-staging area for millions of waterbirds traveling the Pacific and intermountain flyways. As such, Great Salt Lake and its wetlands have been recognized as an essential component of the Western Hemispheric Shorebird Reserve Network. Yet, this valued resource is at potential risk of degradation and conversion due to rapid urbanization and point and nonpoint sources of nutrients and toxics. Farmington Bay wetlands, are receiving the great majority of secondarily treated sewage from a populace of more than one million people along the Wasatch front.

The Great Salt Lake is the fourth largest terminal lake in the world. On average, the lake is 3 to 5 times the salinity of the ocean. The lake is extremely shallow (maximum depth = 37 feet or 11 m). On average, the lake rises 1.5 feet each spring and looses that amount throughout the rest of the year through evaporation. At the average surface elevation of 4200 feet, the lake covers about 1,700 square miles (Figure 1.1). At the historic low surface elevation of 4191.5, measured in 1963, the lake covered only 950 square miles. The drop of 8.5 feet in elevation resulted in a loss of 44 percent in surface area. During 1986 and again in 1987, the lake reached an elevation of 4,211.6 feet and had a surface area of about 3,300 square miles. Hence, depending upon lake elevation, the lake may contain hundreds of square miles of saline mudflats. In other words, for every foot the lake rises or declines, 45,000 acres of mudflats are inundated or exposed. On average, these lacustrine mudflats include about 450,000 acres that have varying degrees of salinity and plant community development, depending upon lake elevation and the time that they are exposed to the flushing effects of rain and tributary flows of fresh water.

About 150,000 acres of wetlands occur in Farmington Bay, occupying the transition zone between the freshwater sources of the Jordan River, other small tributaries and POTW discharges and the hypersaline pelagic waters of Farmington Bay. These fringe wetlands receive and assimilate nutrient inputs from the most densely populated region of the lake's watershed. Since the drought that began in 1999 (resulting in the surface elevation of 4193 by 2004), we discovered that about three years of leaching is required before plants begin to germinate successfully and wetland communities expand. As the region recovers from the drought (average lake surface elevation approximately 4196 ft. in 2006), tens of thousands of acres of emergent plant communities develop each year. Ultimately, the persistence and fate of these temporary plant communities and the concomitant invertebrate and bird communities that utilize and depend on them are subject to annual and long-term average precipitation in the watershed. The majority of the fringing wetlands of Farmington Bay fall into two classes: 1) impounded, defined by human-made levees that were constructed to create large, shallow ponds ranging from about

Figure 1.1. Great Salt Lake images during high water of 1988 (Left) and low water (2002). Note surface area of Farmington Bay (lower right corner of the lake).

20 to greater than 500 acres. These impoundments, constructed to attract and support waterfowl, are located in the delta of the Jordan River (like those of the Bear and Weber rivers); and 2) sheet flow which are created by water releases onto the mudflats from the final (downstream) impoundments, from POTW effluents that discharge directly to the lake (primarily into Farmington Bay) and from several small uncontrolled tributaries such as Kays Creek, Farmington Creek and Davis Creek. Farmington Bay receives POTW discharges from seven major plants either directly or from four plants that discharge to the Jordan River. This causes the Jordan River to be an effluent-dominated stream that contains between 1.5 and 3 mg P L^{-1}. The Jordan River is currently on Utah's 303(d) list for low dissolved oxygen and elevated phosphorus. As the Jordan River approaches Farmington Bay it is carefully controlled and distributed among and through large shallow impoundments (impounded wetlands) that are owned and managed by Utah Division of Wildlife Resources or several private duck clubs. These large ponds have residence times of several days to weeks and salinity increases as waters move through successive impoundments and approach the lake. Outlet water from these ponds and the POTW discharges flow across mudflats (sheetflow wetlands) until it reaches the standing water of Farmington Bay.

Resource managers and conservation groups are concerned about elevated nutrients in these impoundments, sheetflow wetlands and in the open water of Farmington Bay. In response to these concerns, the Division of Water Quality applied for Wetlands Protect Grants from EPA and, beginning in 2004, the Division received three successive grants with the primary objective of developing assessment methods that can be used to establish site-specific water quality criteria for nutrients. These criteria will then be used to determine whether Farmington Bay wetlands are supporting their beneficial use of support for waterfowl and shorebirds and the aquatic life in their food chain. In addition to the EPA grants (totaling $370,000), Central Davis Sewer District contributed $640,000 and the Division contributed $85,000 in matching funds toward this goal. Specifically, our studies described here are directed at understanding basic ecological functions, sensitive processes or species that occur in these wetland systems and how they respond to nutrient and salinity gradients.

As with biological monitoring and assessment goals in streams and lakes, these studies have been designed to 1) identify thresholds of adverse biological or ecological changes to gradients in nutrients and other parameters, such as extreme swings in pH and DO, that are typically associated with hypereutrophy, and 2) identify sensitive and ecologically important responses to nutrient enrichment in Farmington Bay and its wetlands. An array of these metrics would then be incorporated into an index of biological integrity (IBI) that quantifies (scores) various ecological functions against a gradient in nutrients. Ultimately, thresholds along this scoring range will be used to establish beneficial use support status. This effort represents one of the first attempts by any state to establish methods for wetlands 305(b)/303(d) assessment.

Several contractors were hired from academia and consulting companies to perform sample and data collection, laboratory analysis and statistical analysis. Individual reports were prepared by: CH2MHill; Dr. Larry Gray of Utah Valley State College; Dr. John Cavitt of Weber State University, Dr. Sam Rushforth of Utah Valley State College, Dr. Wayne Wurtsbaugh of Utah State University and Mr. Leland Meyers of the Central Davis Sewer District (Appendixes A.1, A.2, B, C, D.1, D.2, D.3,D.4, D.5, E and F respectively). The objectives of this report are to synthesize and summarize data from the various contract reports and provide additional analysis and interpretation.

This report focuses on our investigations of impounded and sheetflow wetlands. A forthcoming report will focus on the open water of Farmington Bay and limnological characteristics and ecological relationships associated with the nutrients and salinity of the open water.

Additional reports addressing the aquatic chemistry and toxicity of selenium in the wetlands will be completed at the end of 2007.

2.0 Methods and Study Design

The initial wetlands study design focused on measuring nutrient attenuation along a longitudinal gradient landward out to Farmington Bay or Great Salt Lake as water passes through successive impoundments or at increasing distances across the mudflats from POTW discharges (Figs. 2.1.1 and 2.1.2). Three or four sampling sites were established along each of these longitudinal gradients. In this manner we expected to describe a co-located biological response along the expected nutrient gradient. Our assumption was that we would observe a systematic attenuation in water column nutrient concentrations (a gradient) at sampling sites located at increasing distance from source waters. Concurrently, we assumed there would be an apposing gradient such that salinity would increase with increasing distance from source waters. In reality however, we discovered that a defined nutrient gradient only occurred at the Ambassador Duck Club. Retention times in the Ambassador ponds were much greater than the other impoundments in this study.

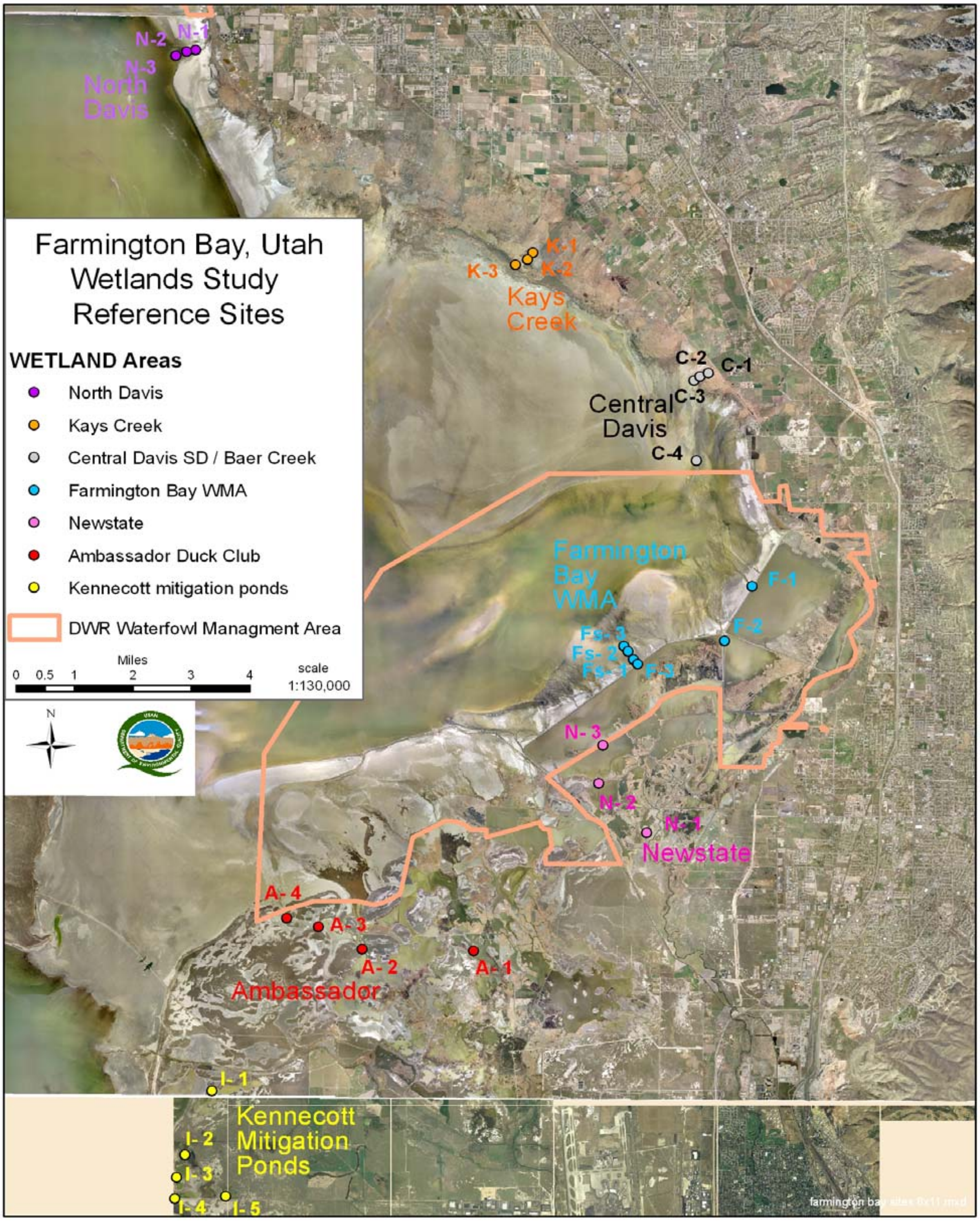


Figure 2.1.1. Sampling sites in Farmington Bay wetlands.

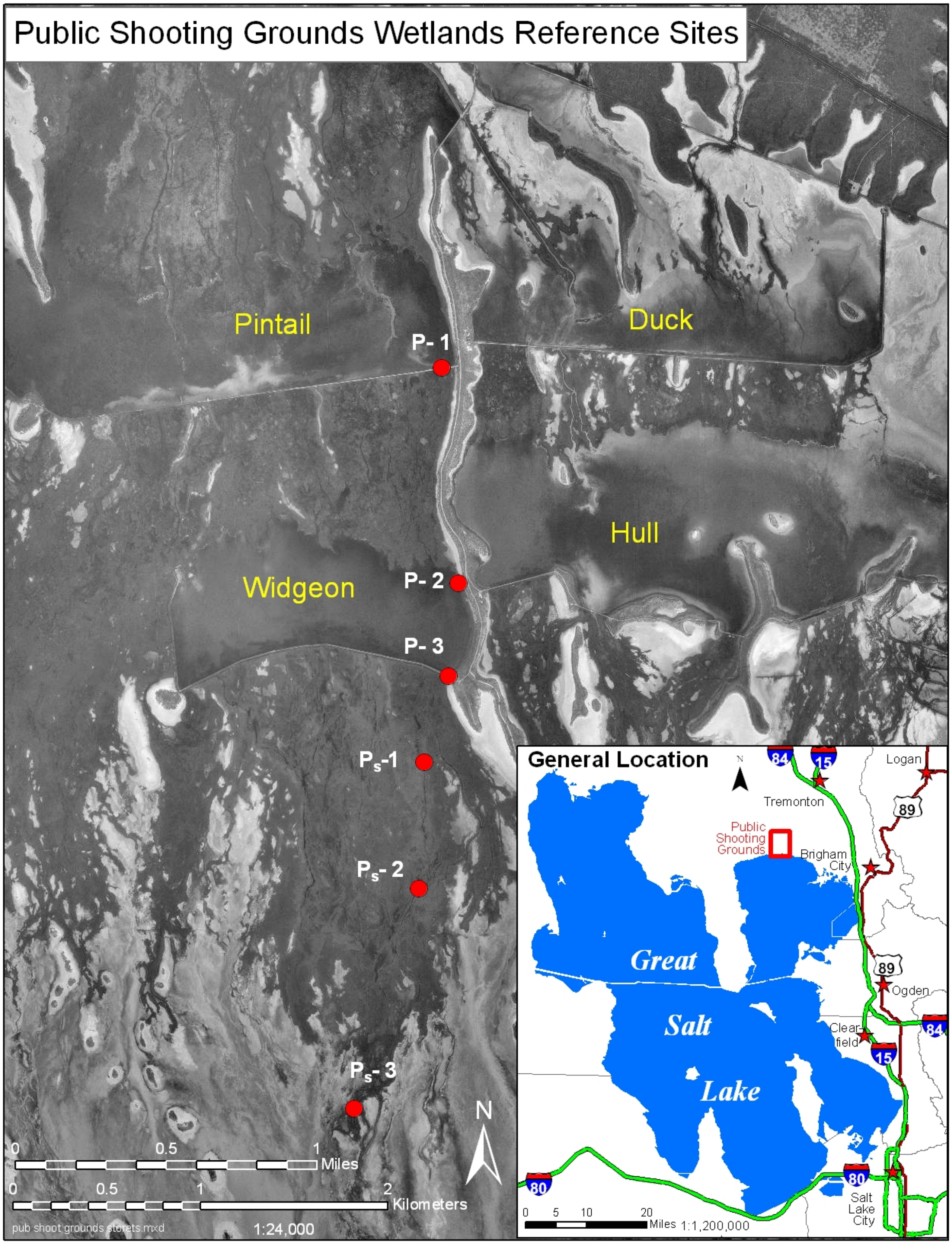


Figure 2.1.2. Wetland reference sites located in the Public Shooting Grounds Waterfowl Management Area in Bear River Bay.

Indeed there were long periods of time when no releases from these ponds occurred and this greater residence time likely resulted in greater assimilation of nutrients. Throughout the remainder of the locations, there was only a slight decrease in water column phosphorus concentrations as the water flowed toward the lake.

Biotic parameters included various macrophyte measures, such as percent aerial cover, stem height, species composition, tissue nutrient concentrations and ratios, and above ground biomass; phytoplankton and periphyton community structure; macroinvertebrate community composition; and phytoplankton and periphyton community composition. Abiotic factors in the water include total P, nitrate-nitrite, ammonia, metal concentrations, pH, EC, dissolved oxygen and temperature. Soil nutrient concentrations, pH and EC were also measured.

The lack of attenuation along our longitudinal transects prompted us to expand our statistical analysis to include factor analysis, which was valuable in identifying general relationships between water quality parameters and various biological response variables. Hence, we used univariate and multivariate analyses, including factor analysis and distance weighted least squares (DWLS). DWLS is a smoothing technique which allows the line to flex locally. Unlike linear or polynomial regressions, which force a line fit to a specific type of equation, the DWLS provides a true representation of the data. Factor analysis using invertebrate assemblages, biological measurements and physical/chemical factors was performed by Drs. Gray, Madon, and Hoven.

In addition, nesting and hatching success of more than one thousand pairs of black-necked stilts and American Avocets were monitored in order to perform a direct measure of beneficial use support. This study included forage availability and stomach analysis to identify food preference and availability. Some nesting habitat characteristics were also noted.

3.0 Results and Discussion

Targeted (nutrient-enriched) sites were identified in the delta area of Jordan River and included Farmington Bay WMA, the Newstate and Ambassador Duck Clubs and the Inland Sea Shorebird Reserve (ISSR). Reference conditions for impounded wetlands were identified at Public Shooting Grounds WMA (PSG) and reference conditions for sheetflow wetlands were identified at sites leading from the discharge point of the final impoundment of Public Shooting Grounds (Widgeon Lake) and at the mouth of Kays Creek. Kays Creek provides water to sheetflow wetlands from a natural (uncontrolled) tributary to Farmington Bay. Although we were careful to find the “cleanest and healthiest” reference sites possible, this proved to be difficult. For example, the Kays Creek reference site experiences considerable urban and agricultural runoff. Phosphorus concentrations routinely ranged from 0.1-0.3 mg L^{-1} total P. Therefore, phosphorus concentrations in the Kays Creek system fell somewhat mid-range between those measured at Public Shooting Grounds (0.02 to 0.05 mg L^{-1} total P) and those measured at the Central Davis Sewer District or North Davis Sewer District outfall (2.6 to 4.2 mg L^{-1} total P.

3.1 Vegetative Community Response

3.1.1 Impounded sites

Submerged aquatic vegetation (SAV) has been shown to be a sensitive indicator of water quality (Kemp et al., 1983; Orthe and Moore, 1983; Stumpf et al., 1999; Tomasko et al., 1996) and sentinel accumulators (Brix and Lyngby, 1983; Burrel and Schubel, 1977; Hoven, 1999; Ward, TJ, 1987; Wolfe et al., 1976) of anthropogenic stressors in shallow estuarine embayments worldwide. SAV provide myriad ecological functions to a watershed. They provide a protective environment and nursery function to invertebrates, fish, and shellfish; stabilize sediments; cycle nutrients and elements; attenuate nutrients and other pollutants; and filter suspended sediments. SAV requires relatively high levels of light and is susceptible to shading by algae (epiphytes, macroalgal mats, and / or phytoplankton), duckweed, suspended sediments, and water color. Increases in algal populations (blooms) are stimulated by increased nutrient loads and often associated with inputs from high human density and / or industrial areas or areas of agricultural runoff (Madden and Kemp, 1996; Staver et al, 1996) and have been shown to correlate with decline in aerial cover of seagrasses (Short and Burdick 1996, Valiela et al. 1997).

It should be pointed out that all of the impoundments along the Jordan River delta are managed for waterfowl support for nesting and fall migration. Because *Stuckenia sp.* is the preferred forage taxa by omnivorous waterfowl, the ponds are managed to optimize SAV growth and, indeed, the submergent plant *Stuckenia filiformis* (fine-leaf pondweed) dominated the impounded sites. *Ruppia cirrhosa* (spiral ditchgrass), another SAV, was present in the late-season samples of the more-saline ISSR ponds and the last pond of the Ambassador Duck Club. *Ceratophyllum demersum* (coon's tail) was also occasionally present in small proportions. There was also a varying amount of floating mats of filamentous green algae (primarily *Spirogyra sp.*), the Cyanobacterium *Oscillatoria* sp. (among others) and duckweed (*Lemna minor*), and epiphytic algae on the SAV at the targeted sites. Although duckweed is somewhat utilized by waterfowl, it is much less preferable than *Stuckenia*, and *Spirogyra* has no known value to waterfowl.

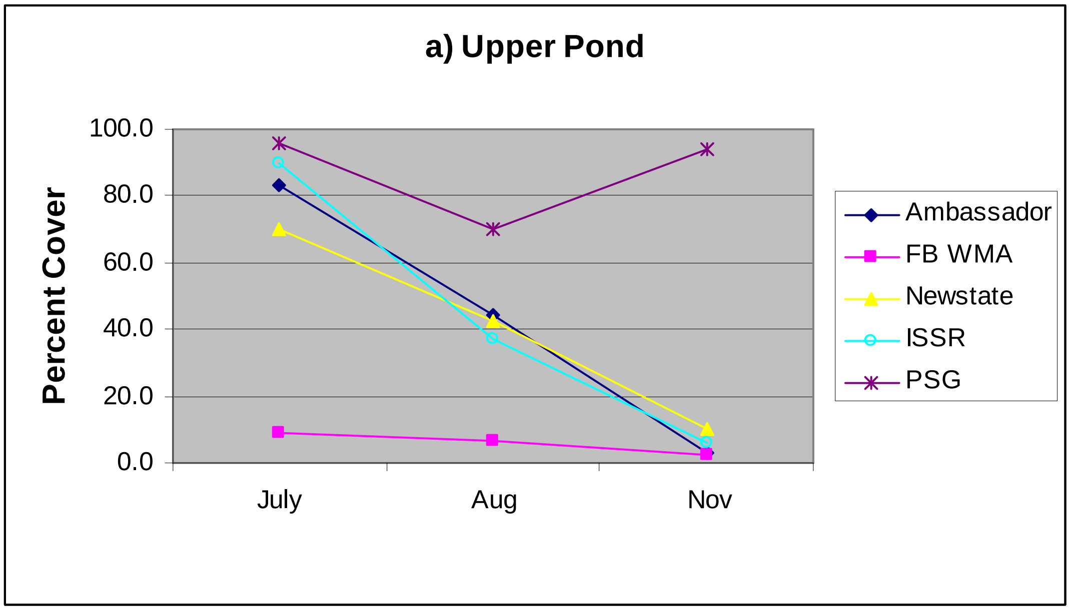
Seasonal biological and water quality sampling revealed substantial differences in plant community responses. The highest percent cover in the upper ponds at the targeted sites occurred in June and July and declined substantially from August through November (Figure 3.1.1a). Percent cover also varied dramatically between seasons at Newstate, Ambassador, and ISSR. Although the ISSR is specifically managed for shorebirds, its "upper pond" (South Pond B) is deep enough to grow *Stukenia* and attract waterfowl.

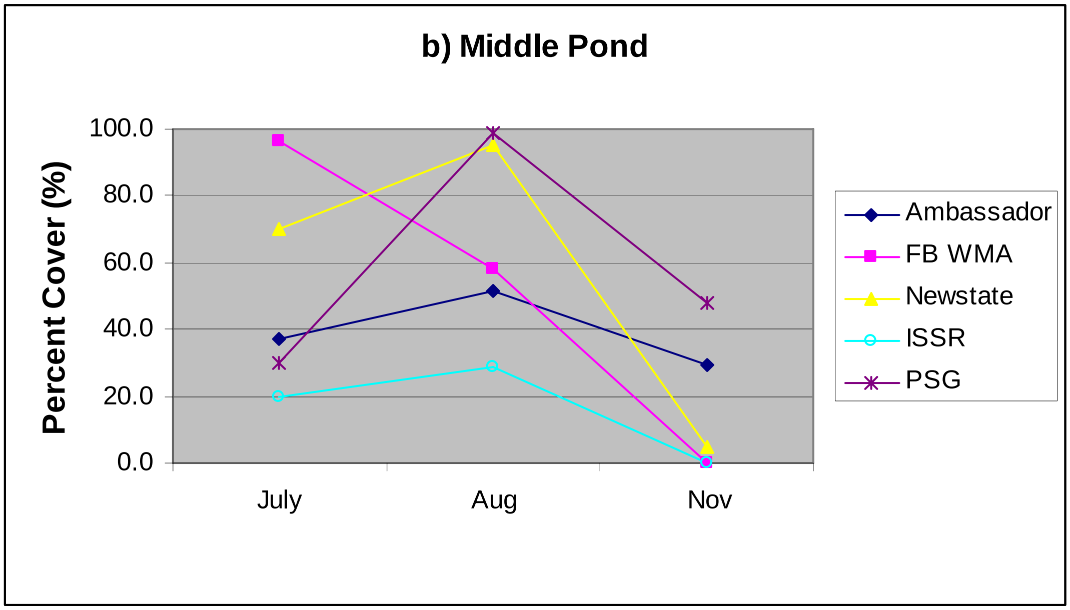
The middle ponds exhibited a hysteretic response in productivity (Figure 3.1.1b). With the exception of FB WMA, percent cover was 70 % or lower during July and increased during August. Percent cover in all ponds declined by November. The calcareous green alga, *Chara* sp., out-competed *Stukenia* for space at PSG middle pond and was the initial cause for low *Stukenia* cover during July. As the summer progressed, *Chara* was not observed and *Stukenia* became strongly established. As with all the impounded areas (except at the ISSR), carp removed a lot of vegetative material in search of macroinvertebrates. At PSG, carp grazing activity in conjunction with grazing pressure from waterfowl was likely the primary cause of reduced SAV cover. Yet, percent cover at

PSG was 2 to 4 times greater than that at the target site middle ponds during November, indicating that additional stress(s) other than grazing may be present at the target sites.

Percent cover SAV showed high percent cover at target site lower ponds at all but Ambassador during July and August (Figure 3.1.1c). The lower pond at PSG had 70 -80 % cover of *Chara* during these months so SAV cover was low in that pond. SAV in FB WMA, Newstate, and ISSR lower ponds declined in cover by November, while that in Ambassador rebounded somewhat. SAV in PSG, on the other hand, increased by November to comparable levels of the middle pond and had almost 3 times as much percent cover as the target sites.

It is possible that this seasonal decline in cover at target sites is a result of heavy grazing by waterfowl as they begin to congregate during mid- to late August. This would be particularly true for Unit 1 of the FB WMA whereby it is managed as a waterfowl resting pond and hence no hunting is allowed. Various waterfowl species readily learn this and congregate in great numbers in Unit 1. Thus, it might be expected that foraging activity would be reflected by the SAV percent cover data as vegetation is intensively uprooted and consumed. However, the Public Shooting Grounds are also managed for waterfowl where SAV did not decline in cover at all (upper pond) or as much (middle and lower ponds) as that at target sites and declines in percent cover of target site SAV was observed before large populations of birds arrived (SAV decline in the upper ponds was observed well before early to mid- September when waterfowl densities are greatest). Summer and fall water quality factors were determined following the factor analysis methods outlined in Madon, 2004 and 2005 (Appendixes A.1, A.2). Of the eight parameters used, TSS, conductivity, and temperature explained the least amount of variability when ordinated in the second and third factors and hence, were excluded to reduce the data to one factor. All water quality data were transformed by Log10, (Log10 $(x + 1)$ for zeros). All % cover data were transformed by arcsine$\sqrt{x}$, arcsine(square root((0+3/8)/(15+3/4)), for zeros after Anscombe (1948). When % cover of SAV is compared with a water quality factor by season (summer vs. late fall) using regression analysis (analysis of variance), most of the impounded sites of this study showed moderate to abundant % cover SAV in the early through late summer and there was no significant difference among sites ($p = 0.364$). By the fall, most sites showed a decline in % cover with increasing nutrients and DO, with the exception of Public Shooting Grounds reference ponds, which had significantly higher % cover SAV than the other ponds (Figure 3.1.2, $p = 0.144$). This occurred even though water column P in PSG remained very low (circa 0.02 mg L^{-1}; and water column concentrations in the target impoundments remained > 0.2 mg L^{-1}).


a) Upper Pond
Percent Cover
100.0
80.0
60.0
40.0
20.0
0.0
July
Aug
Nov
Ambassador
FB WMA
Newstate
ISSR
PSG


b) Middle Pond
Percent Cover (%)
100.0
80.0
60.0
40.0
20.0
0.0
July
Aug
Nov
Ambassador
FB WMA
Newstate
ISSR
PSG

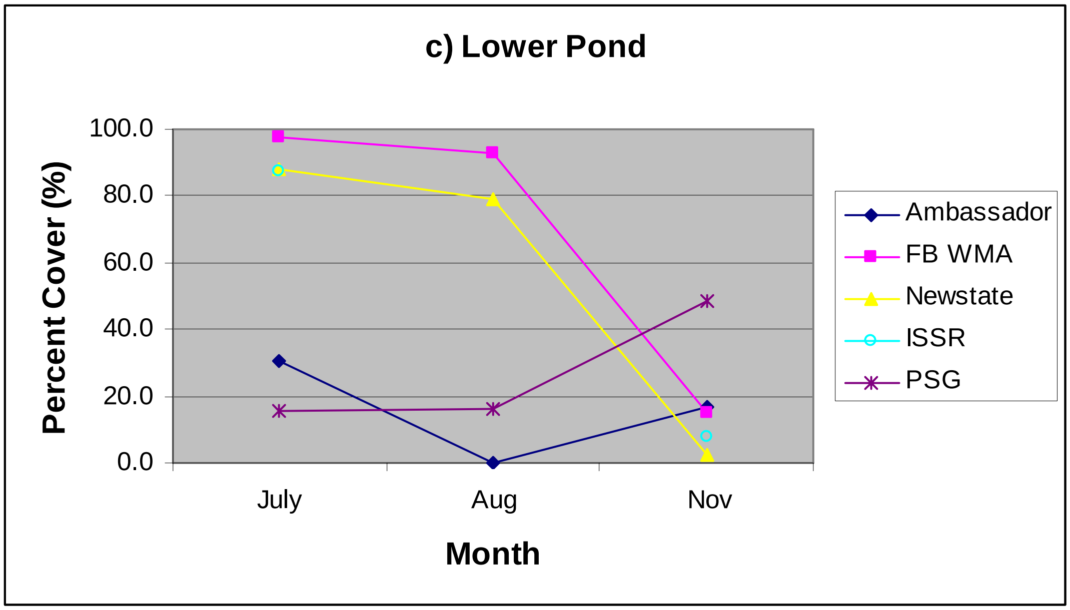


Figure 3.1.1. Seasonal changes in percent cover of SAV for the (a) upper, (b) middle, and (c) lower ponds of our reference system (Public Shooting Ground) and three target systems.

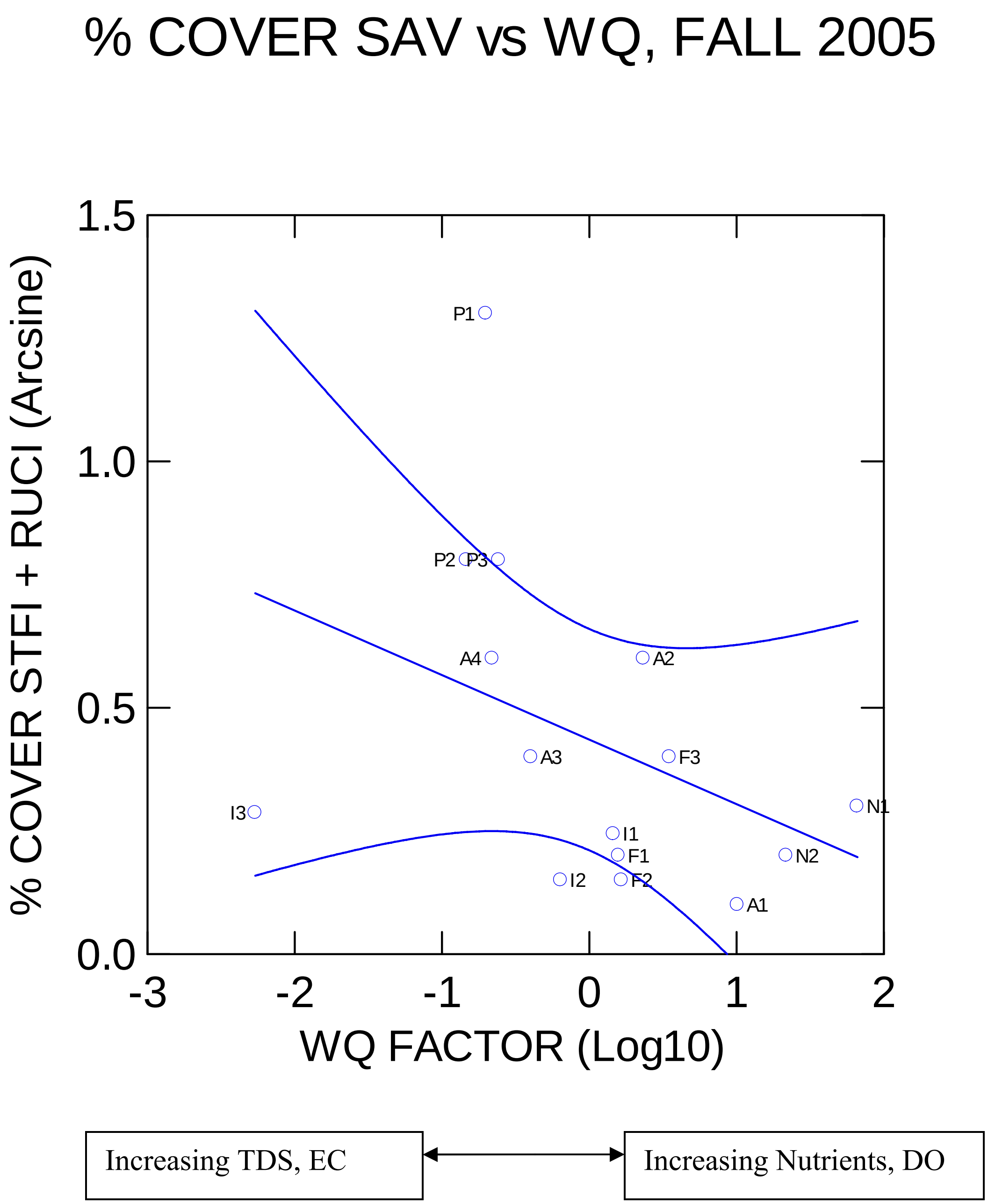


Figure 3.1.2 Percent cover of dominant SAV (*Stukenia filiformis* and *Ruppia cirrhosa*) versus water quality factor at target and reference ponds during the fall of 2005, p = 0.144 ± 95% confidence interval.

This might appear to contradict the paradigm that lower nutrient concentrations should result in less biomass and more nutrients should support greater biomass. However, there are two important factors that support this apparent contradiction:

1) Both emergent and submergent vegetation can derive all of their N and P requirements from sediments (Thiebaut and Muller 2003, Carr and Chambers 1998, Madsen and Adams 1988, Carignan and Kalff 1980). Indeed, Canfield and Hoyer 1988 and Peltier and Welch (1969) found no relationship between macrophyte growth and water-column P and N concentrations. Carignan and Kalff (1980) reported that nine common species of aquatic macrophytes, including *Stuckenia pectinatus*, took all of their phosphorus from the sediments when grown in situ in both a mesotrophic and a mildly eutrophic bay. Even under hypereutrophic conditions, the sediments contributed an average of 72 percent of all the phosphorus taken up during growth. Therefore, submerged macrophytes in PSG obtain adequate nutrients from their associated sediments regardless of nutrient levels in the water column.

2) There is considerable evidence that the early senescence and loss of % cover in the target impounded sites and particularly in comparison to the reference ponds at the Public Shooting Grounds are the result of degraded water quality and related effects rather than normal seasonal changes. Total water column P in the target impounded sites was consistently more than an order of magnitude greater than in the PSG ponds and may be the driving factor that is overwhelming those systems. Often heavy epiphytic biofilms (including sediment) were observed on the leaves of the SAV, and floating and entangled mats of macroalgae (Chlorophyta and Cyanophyta) and duckweed were frequently present at the target sites where nutrients were elevated. The "premature"- senescence of SAV was likely induced by shade-related stress to the SAV by the epiphytic and macro-algal communities, and duckweed in some cases. Additionally, as percent cover of the SAV declines, suspended sediment from the wind events remains in the water column for longer periods since there is no physical structure (i.e. plants) to slow water currents and facilitate settling and water clarity (Short and Short, 1984; Ward et al. 1984). This turbidity causes additional stress on the remaining SAV due to reduced light.

Although somewhat lower, sediment P concentrations in PSG are in the same range as those in the targeted impoundments (see Section 3.4) yet SAV did not show a premature senescence as in the target sites. A simulation of eutrophication responses in submersed estuarine plant communities showed several important responses under nutrient-enriched conditions that may have implications for the Farmington Bay target sites (Madden and Kemp 1996). Epiphytic algal biomass was stimulated by an order of magnitude, while SAV biomass declined severely under both N + P enrichment. Phosphorous enrichment alone has not been shown to trigger community shifts in estuarine production but when N + P enrichments are introduced to mesocosm experiments and model simulations, epiphyte production can be exponential, while attenuating light to deleterious levels to SAV (Madden and Kemp, 1996; Taylor et al., 1995 and 1999). While N levels in most of the target ponds are low to negligible, it may be possible that the observed high occurrence of cyanobacteria provide enough fixed nitrogen locally to support heavy epiphytic growth (Powell et al. 1989). Additionally, N_2-fixing heterotrophs and cyanobacteria associated with duckweed mats have been found to fix as much as 15-20 % of the nitrogen requirement for duckweed (Zuberer, 1982), a substantial amount that could also contribute to the localized water column pool for SAV. The increased density and coverage in duckweed, and filamentous and epiphytic algae in response to increased nutrients has been well documented (Vaithiyanathan and Richardson 1999, Portielje and Roijackers 1995, and others). Accordingly, where nitrogen is limited, rich populations of epiphytic, nitrogen-fixing Cyanobacteria most often accompanies duckweed populations (Duong and Tiedje 1985, Zuberer 1982, Fink and Seeley 1978). This ability to manipulate nutrient availability provides a symbiotic relationship that favors a floating duckweed community. In turn, increased shading and concomitant increased tendency toward anoxia deeper in the water column may severely restrict health and survival of submerged vegetation (Morris et al. 2004). Further, Madden and Kemp (1996) found epiphytic growth on SAV in nutrient enriched scenarios became more dense as leaf tissue area decreased due to leaf mortality and sloughing and was an important factor in the decline of SAV due to increased shading - more so than turbidity related to phytoplankton blooms.

Another important conclusion from Madden and Kemp (1996) was that long-term shading stress to SAV in enriched environments inhibits carbon storage in root and rhizome tissues. SAV roots and rhizomes can provide a root buffering effect such that carbon stored from production periods is reserved for reproduction the following spring. When Madden and Kemp (1996) ran their model for successive years under sustained nutrient enrichment, detrimental epiphyte loads lead to negative P:R (production to respiration ratio) and resulted in reduced SAV biomass, reduced carbon stored in the roots and rhizomes, and ultimately a decreased reproductive potential. They concluded that a “root buffering effect” is essential for long-term survival of SAV beds and to restore plants to historic levels would likely require improvements to water quality that persist for several years to allow root rhizome systems to become re-established.

In Farmington Bay target ponds, it is likely that epiphytic growth on the SAV leaves and presence of algal mats and duckweed attenuated light below critical levels required by *Stukenia* and lead to a premature senescence of SAV. This condition was likely exacerbated as fall progressed and photoperiod and sun angle diminished. During fall collections at target ponds, SAV roots and rhizomes of remaining shoots were often rotting or not well developed (the only exceptions were ISSR T2 (West Pond A) and T3 (Southwest Pond South) and Ambassador T3 (W2) and T4 (W5) where *Ruppia cirrhosa* dominated; Hoven, personal observation). With reduced photosynthetic capacity and resultant reduction in oxygen transport to the roots, below ground tissue may have been susceptible to sulfide toxicity and / or infection by pathogens such as slime mold. Also, germinating seeds were frequently found in the sediment of target sites during the late summer through the fall. On the other hand, plants at the PSG reference ponds grew densely and were difficult to pull (i.e. their roots and rhizomes were well developed and strong) as late as December. Although the plants are perennial, it is likely that roots and rhizomes of SAV at target sites lack carbon stores to regenerate each spring and rely heavily on seedlings each year to maintain the beds.

When C:N:P ratios of similarly aged SAV leaves are compared between target and reference sites during July, all but one target site (Ambassador T2, pond 100) show carbon limitation and all sites (both target and reference) show nitrogen limiting ratios according to Redfield C:N:P ratios of organic matter, 106:16:1 (Table 3.1.1; Redfield, 1934). By late fall, most target sites lacked enough plants to provide enough leaf sample for analysis or were lacking plants altogether. Those that had plants, showed even lower carbon ratios (with the exception of a gain in both carbon and nitrogen above limiting levels at FB WMA T1, Unit 1). Presumably, the improved tissue nutrient ratios at FB WMA Unit 1 during the fall reflect inputs from the pond's use as a rest area for waterfowl; and new SAV growth in grazed areas at that unit may have had less of an epiphyte burden at that time of year and better photosynthetic capacity to fix carbon than during the summer months. However, there was limited SAV cover at that time of year (Figure 3.1.1.a). During the summer, plants at target sites are either competing for carbon with algal and duckweed communities (but not likely since SAV tissue carbon levels are consistent across all sites – see discussion below) or they are not photosynthesizing at optimum capacity due to low light levels a the leaf surface. Although slightly nitrogen limited, reference site plants are better poised to translocate fixed carbon to their roots and rhizomes as they have surplus carbon in their above ground tissue during the summer and fall. Nitrogen is non-limiting at the reference site, PSG T2, and nearly so at PSG T1 during the fall. When the nutrient concentrations of SAV leaf tissue is compared among sites and season, certain patterns come to light. Tissue carbon remains fairly constant at both reference and target sites during the summer and fall (Figures 3.1.3 and 3.1.4). SAV leaf N and P concentrations, however, reflect differences in available N and P levels in the water and sediment. In particular, SAV assimilated high levels of P at the target sites where P is elevated in the water and sediment, and maintained low levels of P at the reference sites both during the summer and fall. When tissue P levels are high, carbon levels remain generally constant, and N levels are low to only moderate, the plants are not functioning at optimum nutrient ratios.

Table 3.1.1 C:N:P of summer and fall SAV above ground tissue for reference and target sites, 2005; n = 3 for all sites. PSG = reference ponds.

SITE	July	November
AMBAS_T1	61:6:1	-
AMBAS_T2	111:10:1	97:9:1
AMBAS_T3	89:8:1	-
AMBAS_T4*	87:8:1	67:7:1
FBWMA_T1	75:5:1	176:19:1
FBWMA_T2	55:4:1	-
FBWMA_T3	79:5:1	58:6:1
ISSR_T1	86:7:1	-
ISSR_T2*	100:8:1	-
ISSR_T3*	73:8:1	-
NEW_T1	72:8:1	-
NEW_T2	90:9:1	-
NEW_T3	76:8:1	-
PSG_T1	166:11:1	188:14:1
PSG_T2	202:14:1	220:16:1^
PSG_T3	205:14:1	161:12:1

**Ruppia cirrhosa*, all other samples were *Stuckenia filiformis*; ^ = collected first week of December; - not enough plant material could be collected for nutrient analysis.

The analysis of the Phase I Farmington Bay SAV % cover data shows a reduction in SAV productivity (expressed as a decline in % cover) in nutrient enriched (target) sites compared to low nutrient (reference) sites. Shading caused by the observed (but not measured) overgrowth of epiphytic and macroalgal communities likely contributed to this low production. In turn, this epiphytic growth may be linked to the elevated P in the water column. A more thorough analysis that would include parameters such as light attenuation across nutrient regimes, biomass assessments of both SAV (as g dry weight · 0.25 m^{-2}) and epiphytes (as chl a), chl a from the water column, and chl a and fluorescence from SAV tissue could better define nutrient and light thresholds below which SAV in the impounded sites can maximize their productivity. Thus, percent cover of SAV may be an important metric for assessing wetland condition and, with refinement, a suite of metrics in SAV communities could be useful in assessing overall wetland condition.

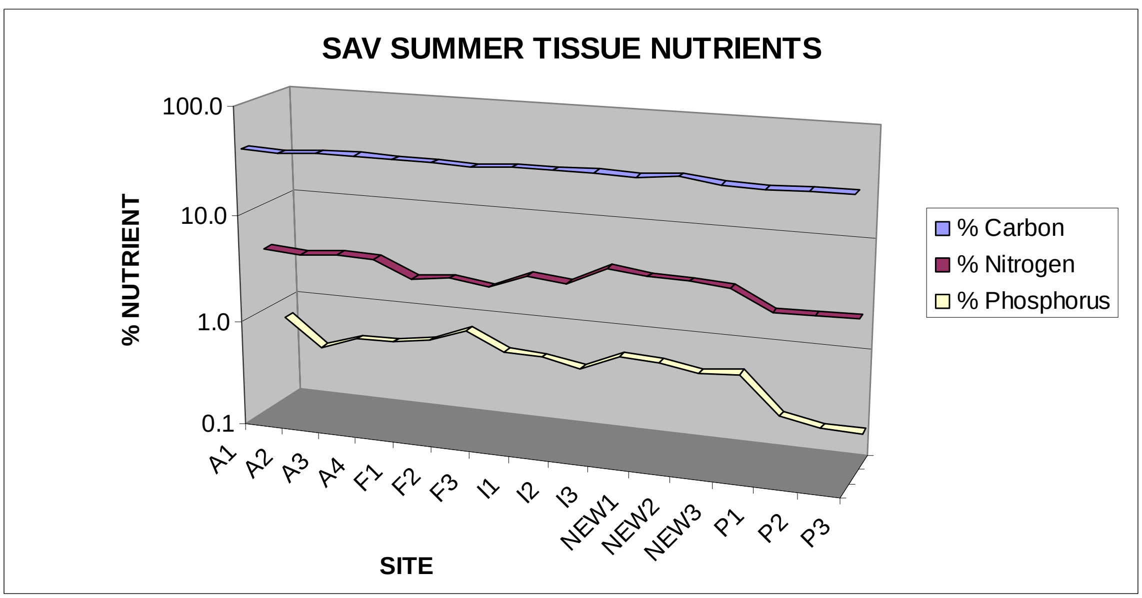


Figure 3.1.3 Percent carbon, nitrogen and phosphorus in SAV summer tissues, 2005. P = Public Shooting Grounds reference ponds.

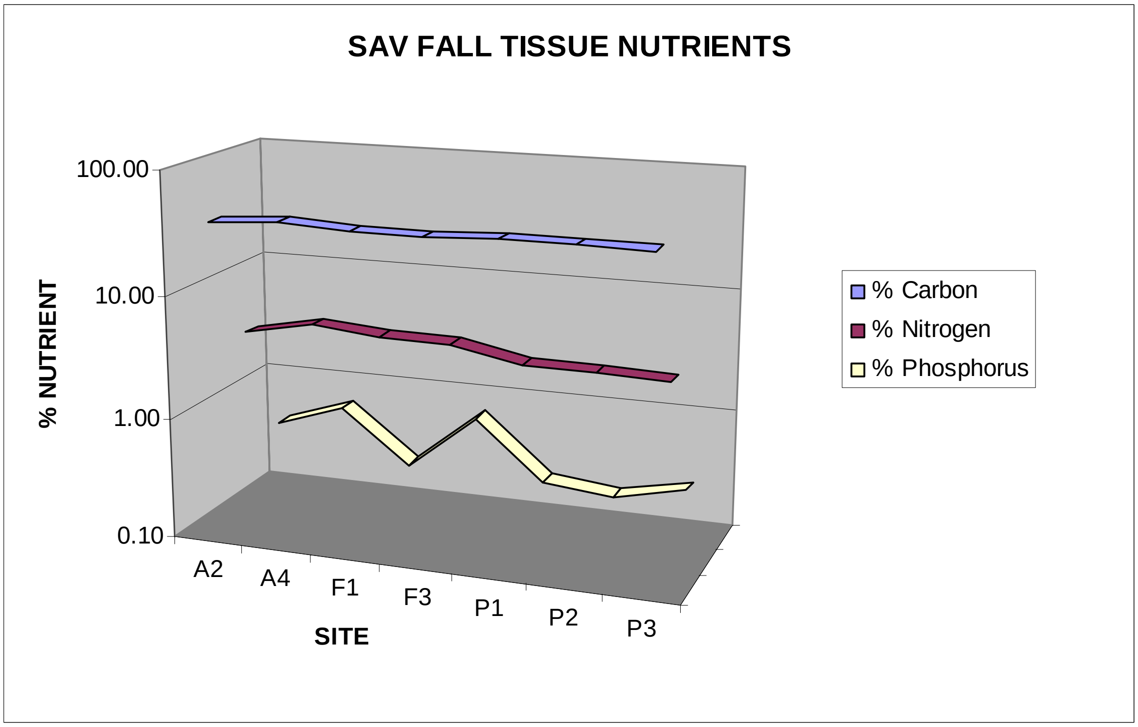


Figure 3.1.4 Percent carbon, nitrogen and phosphorus in SAV fall tissues, 2005. P = Public Shooting Grounds Reference ponds.

3.1.2 Vegetative Community Response at Sheetflow Sites

Statistical analyses of the Farmington Bay data (Appendix A.1 and A.2) assesses trends between a water quality (WQ) factor and percent cover of dominant species (Madon 2004) and community response as presence of native, introduced and invasive species (Madon 2004 and 2005). There are several parameters that were sampled during both years, yet, have not been completely assessed due to time constraints and intuitive focus on key parameters. This summary reviews the results of the statistical analysis and presents a cursory overview of additional relationships and interpretations of the data.

Dr. Madon conducted three tiers of statistical analyses on water and sediment quality parameters versus various plant community measurements from the sheetflow sites and univariate analysis on water quality parameters versus biotic variables (Appendix A.1 and A.2). Many measurements of the plant community were inversely related to water and soil pH. These included *Typha* and *Phragmites* % cover, and *Scirpus americanus* and *Distichilis spicata* stem height. A list of general conclusions summarizes these preliminary results (Appendix A.2; page 26 and 27.) The 2005 spring and summer data were initially combined for analyses (Appendix A.2). This seasonal data will be re-analyzed separately because understanding system function often depends upon temporal trends.

Water quality and water quantity are central to understanding community responses within emergent wetlands of Farmington Bay. Nitrogen (NO_3 and NO_2) rapidly attenuates with distance from the source (except at NDSD where, because of the large volume and short distance to the open water, N remains high along all three sampling sites, Section 3.4). Phosphorus, on the other hand, is often maintained at the same (or in some cases, higher) levels with distance from the source waters. Salinity increases with distance from the POTWs and WMAs.

At the first two transects of CDSD (C1 and C2), there is consistent, high flow rate and high nutrients (low WQ) and low number of native yet invasive species of plants (*Phragmites australis* and *Typha latifolia*). As the effluent moves through these areas, nutrient uptake by plants and its resultant nitrogen attenuation in the water column should be reflected in the primary productivity of the plants (as above ground biomass (AGB)) until increasing salinity begins to limit growth of non- salt tolerant species. Unfortunately, however, there was a quality control issue with the 2005 AGB samples at a subcontracted laboratory and the data from all sites was lost. When the flow rate dissipates by the third transect (C3), the number of native species increased but many of the species have invasive tendencies. The fourth transect (C4) is much further from the POTW (approximately 2 kilometers), and salinity is elevated. There is a decrease in the number of native species at this point and none of them are invasive. There was a moderate reduction in nutrients at C4. Similar community responses of vegetation along flow and salinity gradients occurred at NDSD.

Total number of native species from each site was plotted against the water quality factor (Fig. 3.1.5). Transects C1 and C2 fall at the negative end of the water quality factor with low species diversity including native invasive species and low water quality (high nutrients) yet the span of wetlands that the transect data reflects provides for N attenuation in the effluent. Phosphorus, on the other hand, does not attenuate as the effluent passes through these and subsequent transects – suggesting that P absorption by plants in this system is maximized. Yet, the threshold of maximum P assimilation by wetland plants in this region is not well understood, however, further discussion on this issue is presented in Section 3.5 below.

NDSD transects 1 (N1) and 2 (N2), C3, Kays Creek transects 2 (K2) and 3 (K3) fall in the mid-range of the water quality factor and may be showing a threshold response. Under certain conditions: eg. moderate flow, encompassing a sediment deposition zone, moderate to moderately high nutrients, and in some cases, other disturbance (eg. cattle, four wheeler activity), the number of native species increased, but the proportion of invasive species is also high. The increased proportion of invasive species is indicative of an imbalance in the system where elevated nutrients and other disturbances such as erosion and sediment deposition allow for their proliferation. It is not clear at what level nutrients and other disturbances trigger proliferation of invasive species. Thus, this mid-range of the water quality factor may be showing a tipping point leading to disturbance-based community responses (a fulcrum), or an apex of maximum disturbance. At the positive end of the water quality factor, FB WMA transects 1 (Fs1) and 2 (Fs2) and Public Shooting Grounds (PSG) transects 1 (Ps1), 2 (Ps2) and 3 (Ps3) showed reduced number of native species, reduced number of invasive species and moderately high to high water quality.

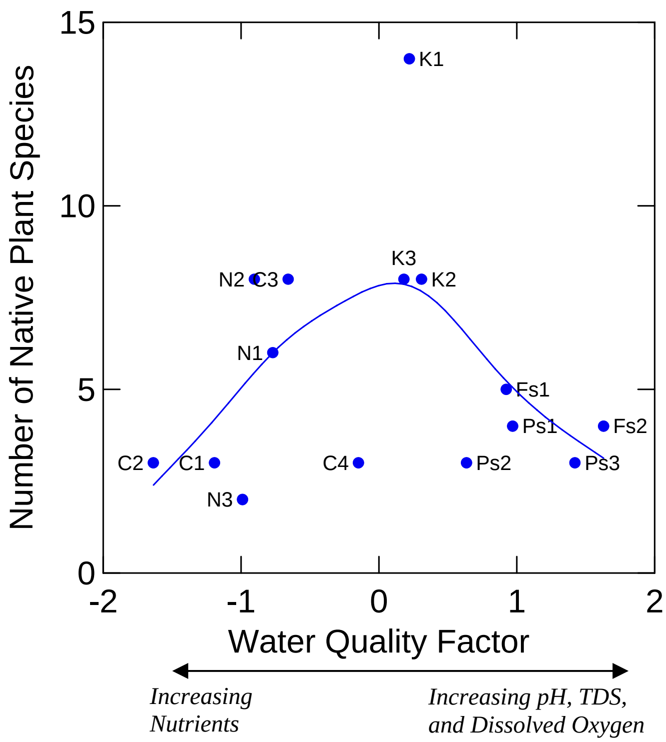


Figure 3.1.5. Total number of native species versus water quality factor at sheetflow sites, 2005. (From Maden, 2005; Appendix A.2)

FB WMA sheetflow sites have moderate levels of salinity (measured as electrical conductivity, EC, and total dissolved solids, TDS) and PSG sheetflow sites are more saline compared to other sites, so it is not clear whether salinity alone is responsible for limiting the occurrence of some invasive species found elsewhere.

There were no sites with nutrient levels between those of Kays Creek and Fs1, Ps1 and Ps2 and that had low EC or TDS so we are unable to exclude salinity as the only factor that kept invasive species in check. There may be conditions where natural competition from native, non-invasive species is not compromised, but those conditions do not appear to be described by the current data.

Unlike the impounded sites, plant tissue analysis in sheetflow sites indicated that nitrogen is limiting at only a handful of sites (FB WMA and KC T1 and T3 during the early summer, and CDSD T1 and T2, KC T1 and NDSD T1 during the late summer, Table 3.1.2). Because N:P ratios vary more among sites than among species, it is possible to compare sites for limiting nutrients using various species (Güsewell and Koerselman, 2002). It is clear that plants at CDSD, NDSD, and PSG are not N limited with any distance from the source water during the early summer (gray shaded ratios). At PSG, there are many springs within the sheetflow area that supplement the outflow from the impounded outlets and may prove to be sources of nitrogen for the plants. By the late summer, plants at CDSD T2 and T3, and NDSD T1 became N limited, while all sites at FB WMA were not N limited.

Nutrient concentrations of the same-aged leaves of the dominant species at each site are shown in Figure 3.1.6 (early summer) and Figure 3.1.7 (late summer). Variations within sites that have a shift in dominant species with distance from source waters (and increase in salinity) may reflect different assimilation capacities between species (Güsewell and Koerselman, 2002). However, plants at sites with either high N or high P generally reflected high water nutrient concentrations in their tissues (see Section 3.4).

Some anecdotal descriptions of the study sites are worth noting. Flow rates differed among sites and could contribute to the various vegetative community responses. For example, NDSD has a much higher volume of effluent than CDSD and although braided channels form, the velocity remains high enough to limit macrophyte growth within the channels. However, considerable deposition occurs between these channels as well as downstream. These areas of higher elevation provide a different environment for wetland plants than within the channels (soils may be saturated but are not inundated all the time), allowing increased species diversity. Yet, the scouring from erosional forces of water and sedimentation would limit species composition to plants that can tolerate that kind of disturbance. CDSD effluent volume is about 1/3 that of NDSD and, rather than form channels, flow rates are slow enough to allow emergent vegetation to develop in the immediate vicinity of the discharge point. Consequently, substantial spreading of the water occurs to form a true sheetflow condition with very few areas of deposition.

Springs were present throughout the PSG sheetflow site and may cause variability in WQ and plant community composition. The hydrology at Kays Creek varies from year to year such that water flowed from the bank in a southward direction during 2004 and continued through the transect areas. During 2005, debris in the main channel diverted flows to the north side of the bank, leaving the original K1 location dry. Therefore, all three sampling stations were re-established on the north side of the main channel in order to capture consistent flows. During 2004, Kays Creek management sprayed the wetlands by air for Phragmites control. The following year showed very little growth of all plant species (even by the early summer) and the longitudinal transect was discontinued.

In addition to factors affecting hydrology and WQ at several sites, we noted two additional sources of disturbance at Kays Creek. Cattle use the area for grazing and likely increase nutrient levels in surface flows and contribute to the observed increase in invasive species (through trampling existing vegetation and the soil, and seed transport via hoofs / hide and manure). There was also a considerable network of four-wheeler trails for mosquito control applications at Kays Creek and somewhat at CDSD for spraying and sampling purposes.

Table 3.1.2 N:P ratios of emergent vegetation at sheetflow sites during early summer (June) and late summer (August / September) of 2005; n = 3 at all sites.

Site	Species	Early Summer	Late Summer
CDSD_T1	TYLA	20:1	18:1
CDSD_T2	TYLA	20:1	13:1
CDSD_T3	SCMA	21:1	12:1
CDSD_T4	SCMA	21:1	17:1
FBWMAs_T1	PHAU	14:1	23:1
FBWMAs_T2	TYLA	11:1	17:1
FBWMAs_T3	TYLA	13:1	22:1
KC_T1	TYLA	13:1	11:1
KC_T2	SCMA	16:1	-
KC_T3	SCMA	15:1	-
NDSD_T1	PHAU	26:1	15:1
NDSD_T2	SCMA	19:1	23:1
NDSD_T3	SCMA	18:1	19:1
PSGs_T1	SCMA	18:1	25:1
PSGs_T2	SCMA	20:1	30:1
PSGs_T3	SCMA	19:1	28:1

Species codes are as follows: TYLA = *Typha latifolia* (cattail), SCMA = *Schoenoplectus maritimus* (alkali bulrush), PHAU = *Phragmites australis* (Phragmites). Gray = N limiting; - sites not sampled. Note species vary within some sites and among sites.

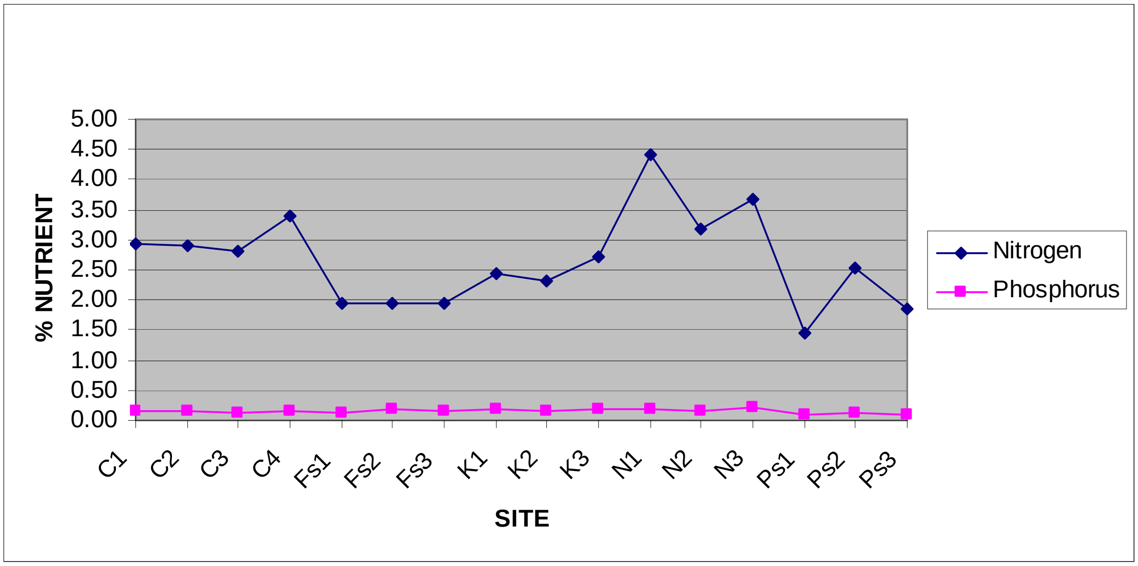


Figure 3.1.6 Percent tissue nutrients (nitrogen and phosphorus) in emergent leaves, early summer 2005; n = 3 at all sites.

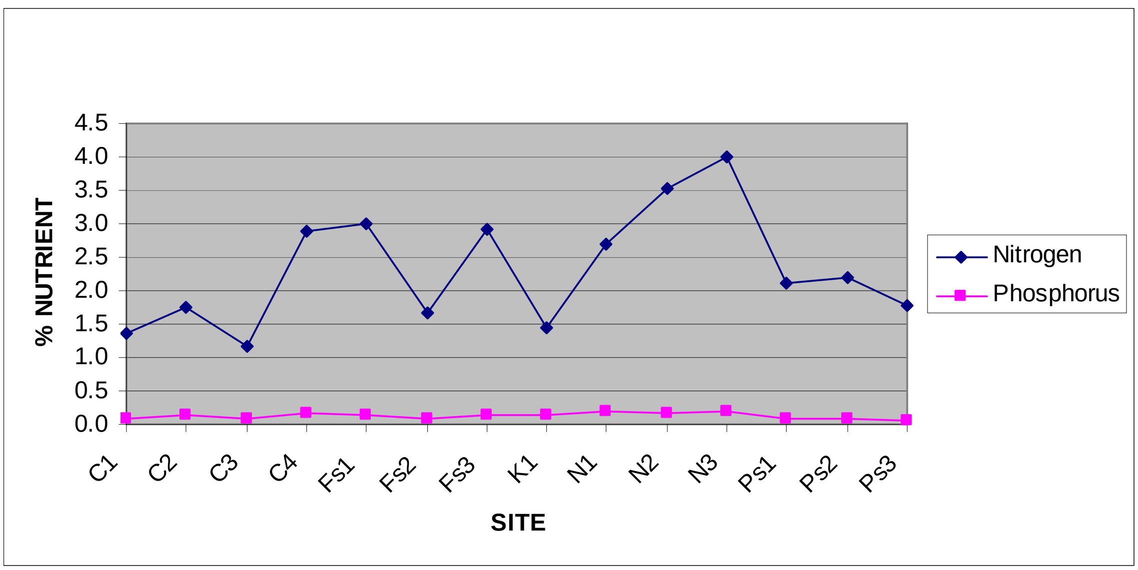


Figure 3.1.7 Percent tissue nutrients (nitrogen and phosphorus) in emergent leaves, late summer 2005; n = 3 at all sites.

3.1.3 Summary of Data Gaps

Additional understanding of the ecosystem and refinement of metrics can be achieved if some of the gaps identified in the data are addressed. Results of our initial efforts at impounded sites indicate that SAV can be useful indicators of stressors in their sub-watersheds and development of metrics based on parameters describing the SAV community shows good potential. Additionally, detailed measurements of nutrient loading, DO, light attenuation and turbidity and other known stressors (e.g. metals) in relation to SAV biomass and % cover may isolate and identify nutrient and turbidity thresholds of SAV impacts, including poor production, premature senescence and loss of below ground carbon stores for future re-growth.

At sheetflow sites, further assessment of biomass relative to nutrient loading and assimilative capacity of nutrients by emergent species will help define the condition of the wetland and its sub-watershed. Additional research needs to focus on identifying potential stressors at low to moderate energy sites i.e. anoxia, sulfide toxicity, nutrient loading, other biological and chemical disturbances versus a high energy system with higher flow rates (a system that is dominated by physical stresses that may either correlate with or confound nutrient related stresses). Although we observed higher above ground vegetative biomass at CDSD (C1, C2, C3), NDSD (N1, N2), and FB WMA (Fs1, Fs2) than reference PSG sheetflow sites, the target sites were composed primarily of Phragmites and cattail versus saltgrass, which also has invasive tendencies but is a much smaller plant. Both Phragmites and cattail are well documented for removing nutrient burdens from water as a form of treatment and are acceptable in performing such functions, However, current loading rates (~ 8-12 g m^{-2}) exceed recommended values (2 -4 g m^{-2}). This brings into question the actual efficacy of Phragmites and cattail to remove nutrients in this situation. Further assessment of biomass relative to nutrient loading and assimilative capacity of nutrients by emergent species would provide a metric for determining whether sheetflow sites outside of POTW and WMAs are capable of treating nutrient enriched water.

The following identifies additional studies that would improve our understanding of these wetlands and provide for additional and potentially important metrics that will contribute to a more complete and accurate assessment of beneficial use support.

3.1.3.1 Impounded

- Effects of light attenuation by epiphytes (with a control for duckweed and macroalgae) on submergent plant community
- Biomass comparisons of SAV communities among sites (as chl a of phytoplankton and epiphytes, and g dry weight SAV/ unit area)
- Turbidity as TSS in the water column (vacuum pumped and filtered portion of the water column)
- SAV as indicators of watershed stressors by assessing Photosystem II fluorescense
- Fall carbon stores in SAV roots and rhizomes in nutrient enriched sites vs low nutrient (reference) sites
- Sulfide toxicity as acid volatile sulfides (AVS)
- Statistical interaction between grazing activities from carp and % cover of SAV in nutrient enriched versus low nutrient (reference) sites
- Differences in grazing pressures from waterfowl by site and by season

3.1.3.2 Sheetflow

- Flow data and influence of velocity and channel depth on vegetative community structure
- Sedimentation rates and identification of deposition zones
- Are sulfates / sulfides overwhelming the oxidizing capacity of roots / root zones (AVS analysis)
- Relationship of AGB (and plant height and % cover) to soil and water nutrients
- Continued literature research on tolerance to various stressors (nutrients, velocity, suspended sediments, salinity, heavy metals, etc.) by species
- Nutrient assimilation capacity of wetland plants (empirical and literature research)
- Freshwater, low nutrient response of wetland plant community (all metrics)
- Is there a reasonable distance / wetland acreage for various flow rates through wetlands that renders acceptable nutrient attenuation during low lake-level years under current loading conditions?
- Are there management alternatives at POTW and WMA outfalls that might improve the biological integrity of their sheetflow sites?
- Finally, will any wetland or POTW design alternatives or combination thereof sufficiently reduce nutrients to a level that will improve the eutrophic conditions of the open water of Farmington Bay? (i.e. reduce cyanobacterial blooms, elevate DO and increase aquatic life diversity.)

3.2 Macroinvertebrate Communities

Invertebrate sampling was performed during late fall in 2004 and during summer and early fall in 2005 at the sheet flow sites and during summer, early fall and late fall in the impounded sites.

Univariate and multivariate statistics were used. Univariate analyses that were particularly useful include the responses of various taxa to water or soil pH. These include mayflies (Fig. 3.2.1a), corixids (Fig. 3.2.1b) and midges (Fig. 3.2.1c). Notably, abundance of these taxa began to decline at pH values between 9 and 9.5.

In addition to their sensitivity to pH, mayflies exhibited sensitivity to DO (Fig 3.2.2). However, the accuracy of this observation may be suspect. For example, diel DO measurements made among all of the study sites demonstrated that DO dropped to near or below 1 mg L^{-1} in most of these ponds during evening hours. Yet, mayflies were found in all of the Ambassador ponds during all three sampling periods and were occasionally found in the Newstate ponds. Hence, it is possible that this mayfly (*Calibaetis* sp) is more sensitive to pH or perhaps some other habitat parameter that has not been evaluated yet. For example, samples collected in the emergent vegetation along the pond fringes vs the submergent habitat will be performed during 2007.

Factor analysis was also performed on these data sets. When comparing water chemistry with macroinvertebrates, the primary chemical factor included the alignment of increasing pH, conductivity, total P and decreasing dissolved oxygen on the X axis being associated with increasing numbers of chironomids and leeches on the Y axis (Fig.3.2.3). Conversely, decreasing pH, EC, Total P and increasing DO was associated with increasing numbers of mayflies, odonates and hemipterans.

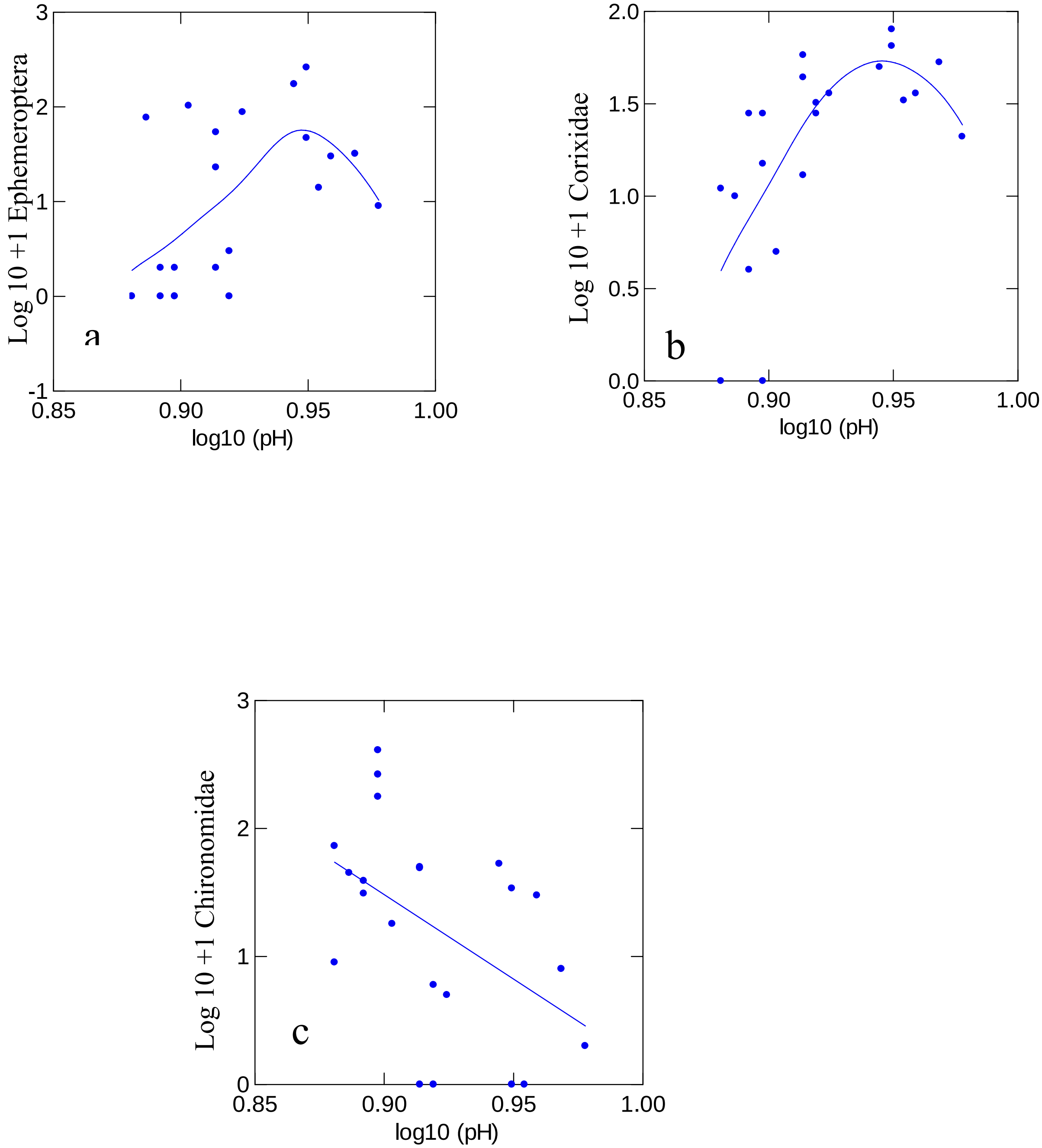


Figure 3.2.1. Responses of mayflies (a) Corixids (b) and midges (c) to differences in pH. For reference, the antilog of 0.96 equals pH 9.1.

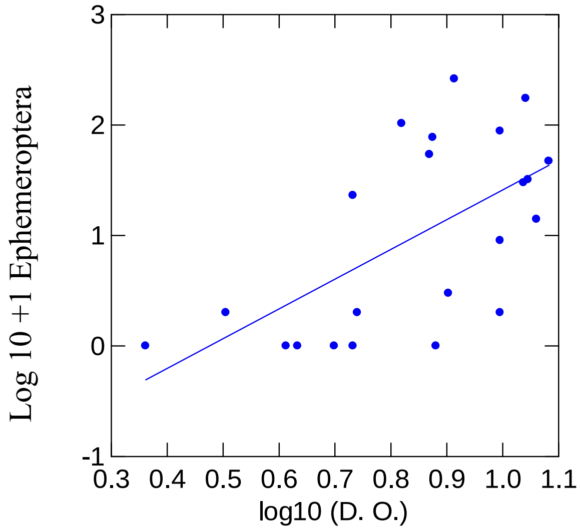


Figure. 3.2.2. Occurrence of mayflies (*Calibaetis* sp) in relation to dissolved oxygen.

The relationship between invertebrate and vegetation factors was also evaluated. Invertebrate communities dominated by mayflies, damselflies, water boatman, scuds (*Hyallela azteca*) and snails occurred at sites dominated by Stuckenia (generally impounded sites) (Fig. 3.2.4a). Conversely, sites dominated by midges, flatworms and leeches occurred where Phragmites, cattails and both Scirpus species were the dominant plant species (generally sheet-flow sites). This was also reflected in the multivariate factor analysis on the invertebrate, vegetation and water quality factors (Fig. 3.2.4b; see Appendix B). Similarly, sites dominated by mayflies, damselflies, water boatman, backswimmers, Hyallela, snails and Stuckenia were relatively more saline and less nutrient enriched. Conversely, the more eutrophic, but fresher sites, were dominated by Phragmites, cattails and both *Scirpus* species and an invertebrate assemblage composed mainly of chironomids, flatworms and leeches.

Also notable, numbers of climbing and clinging (on vegetation) macroinvertebrates declined after the June samples and remained at very low numbers among the targeted impoundments and particularly in FB WMA Unit. It is likely that this reduction in numbers was associated with the decline in SAV that provides habitat. We feel that refinement of sampling strategies, including additional diel measurements of DO and specific local habitat parameters during the 2007 season will reduce sample variability and more accurately assess the importance of local habitat.

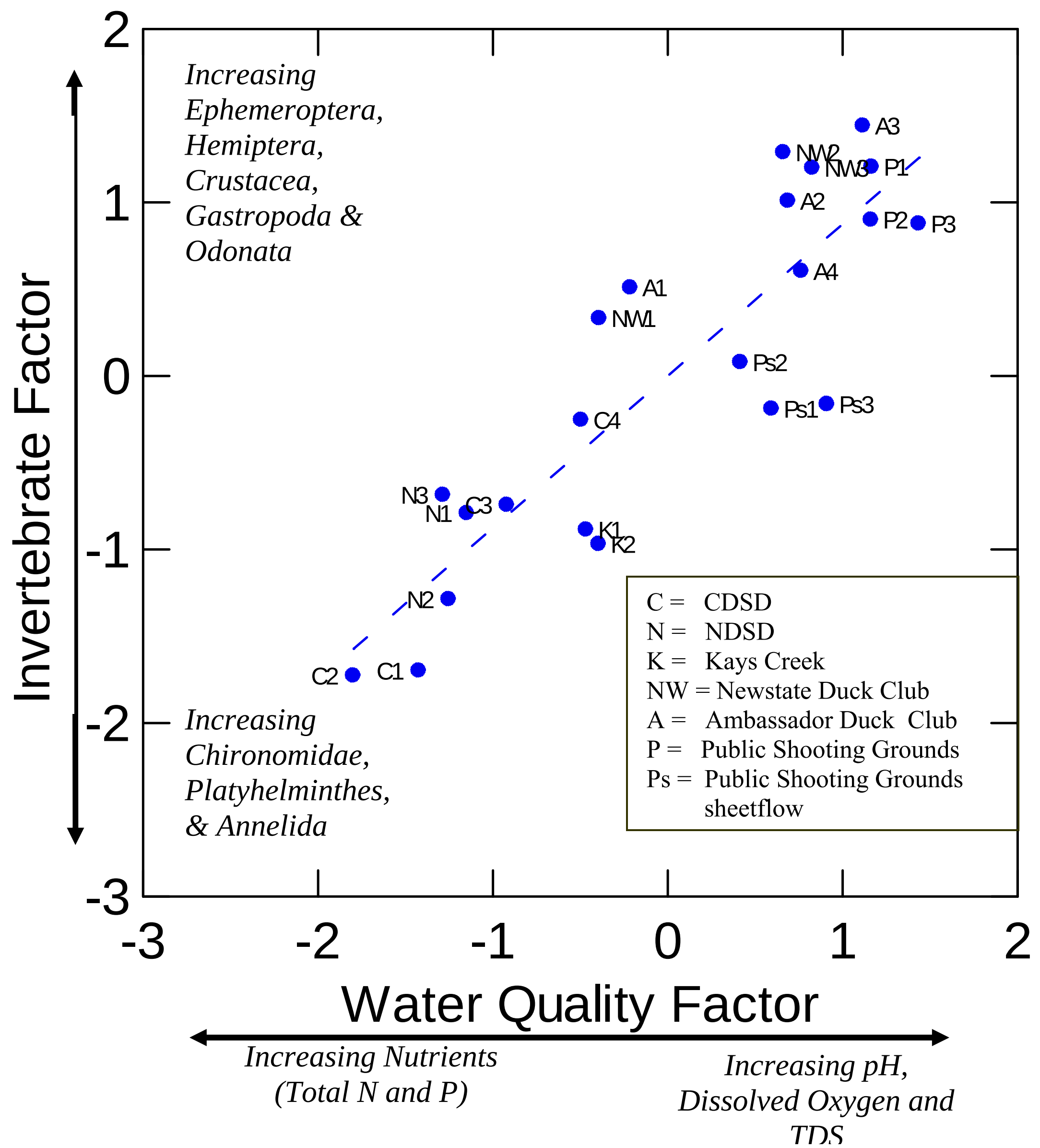


Figure.3.2.3. Results of factor analysis for the primary water quality factor and the primary invertebrate factor. Note general trend toward tolerant species with increasing nutrients.

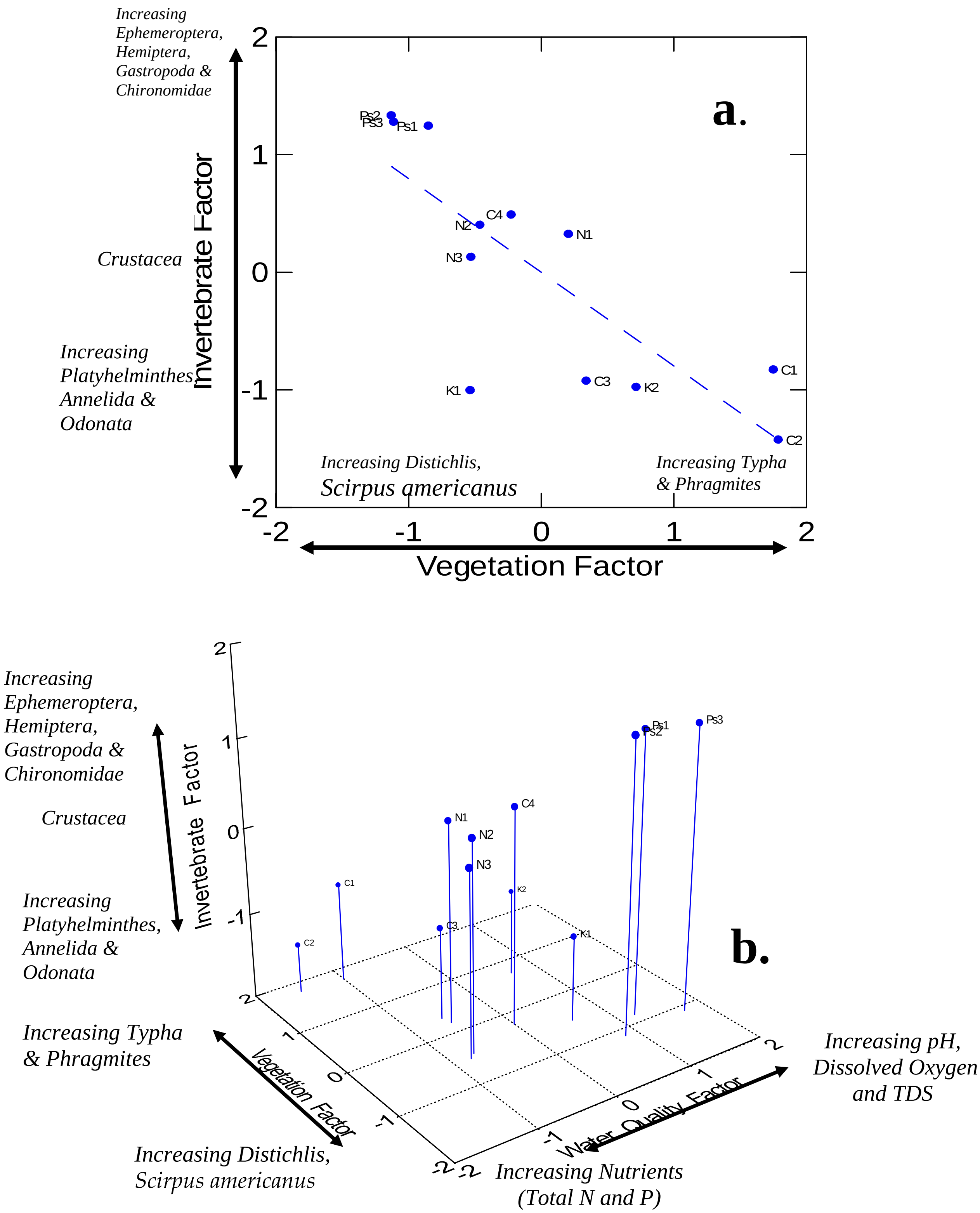


Figure 3.2.4. Results of factor analysis comparing the primary invertebrate factor with the primary vegetation factor (a) and combining the invertebrate, vegetation and water quality factor (b).

Another important caveat to note is that Davis, Weber and Box Elder counties conduct an aggressive mosquito abatement program in an effort to control the spread of West Nile virus. Active spraying from aircraft as well as ATVs was often observed by our field personnel. It is known that both BTI and malathion, a general pesticide used here as an adulticide, are used, depending upon the presence of larval vs adult mosquitoes. It is also known that midges are equally sensitive to BTI as are mosquito larvae and all the taxa present are sensitive to malathion. Location, frequency and pesticides used were actively noted during the 2006 field season and are now being determined for the 2005 field season in order to determine if spraying could have influenced 2005 invertebrate sampling results. Logging the spraying schedule for mosquito abatement will add substantially to the tolerance database and account for the overall influence of pesticide spraying. In addition, during 2007 we will sample macroinvertebrates and water for pesticide analysis to determine whether pesticides are reaching toxic concentrations in the water column and the concomitant invertebrate community structure.

3.3 Shorebird Studies

Shorebird studies were performed to provide a direct assessment of the designated beneficial use for Great Salt Lake (Cavitt 2006; Appendix C). This use has been defined as: "support for waterfowl and shorebirds and the aquatic life in their food chain." Data were collected during 2005 and 2006. These studies were able to incorporate a larger database of nesting colonies that has been supported by a National Science Foundation research grant. This study focused on American avocets and black-necked stilts that were nesting and feeding in the sheetflow environments. Study sites included locations in PSG, Bear River Migratory Bird Refuge, Great Salt Lake Shorelands Preserve (near the mouth of Kays Creek), FBWMA (near our WMA sheetflow water quality study sites and near the Salt Lake sewer canal) and in the ISSR. Farmington Bay WMA and Bear River Migratory Bird Refuge have an active predator control program and this undoubtedly contributed to nesting success and juvenile survival.

Study objectives included a description of nesting habitat and measurement of nesting success, hatching success and prey item selection. Prey selection was determined by collecting individuals immediately after they were observed feeding for at least five minutes and then dissecting out the digestive tract.

Nest site preference included areas with little or no vegetation that provide an unobstructed view by the attending adult. These included areas of early-stage communities of pickle weed (*Salicornia sp.*), or alkali bulrush (*Schoenplectus maritimus)* that were in close proximity (generally < 30 m) to surface water. Close proximity to water is essential in that the young are not fed in the nest. Rather, within 24 hours of hatching, the parents lead the young to surface waters where they begin foraging for themselves. Although these foraging areas include taller vegetation, providing essential cover, the adults attend to the young until flight is achieved.

A summary of nesting and hatching success for both species and for both years are summarized in Table 3.3.1. Hatchability and number of young leaving per nest were consistently between 93% and 96%. These are similar values to those measured in Bear River National Bird Refuge, both of which are equal to or greater than any other success rates reported in the literature.

Birds and macroinvertebrate samples were collected from each of the study sites in order to determine forage availability and forage preference (Appendix C). These data are summarized in Table 3.3.2 and illustrated in Figures 3.3.1 through 3.3.3. The most important invertebrate taxa consumed by the avocets and stilts were Corixidae (water boatmen) and Chironomidae (midges). In fact, 63% of the avocet diet was comprised of just three taxa (Corixidae, Chironomidae and Ephydridae (brine flies)). The black-necked stilt diet was slightly more diverse with 65% of the food material consisting of Corixidae, Chironomidae, Hydrophilidae (water scavenger beetles) and miscellaneous coleopteran (beetle) parts.

With regard to prey selection, the proportion of chironomids consumed by avocets did not differ from the proportion available as identified in the sweep samples. Likewise, there were no differences in the proportion of Corixidae consumed relative to the proportion available. However, the black-necked stilt diet exhibited a slight preference for corixids. They had a smaller proportion of chironomids than were in the sweep nets and a greater proportion of corixids than that in the sweep nets. Cavitt (2006) suggests that this preference is associated with their primary foraging behavior whereby stilts generally peck items from near the water surface and hence are likely attracted by movement. This would favor corixids as they are continuously active (periodically ascending to the surface for air) as compared to the more sedentary and benthic midges.

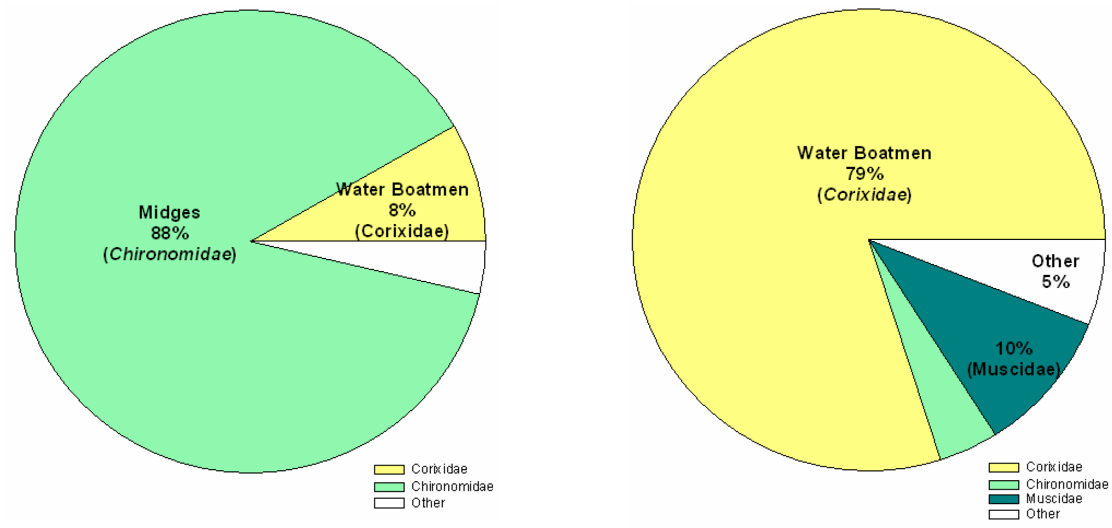
Overall, corixids and chironomids made up the majority of the diet for both species. In view of the known diverse diet and opportunistic feeding behavior of both avocets and black-necked stilts, the preponderance of corixids and chironomids in the diet is likely due to the cosmopolitan occurrence and density of these two taxa among Great Salt Lake wetlands.

Table 3.3.1. Measured values of productivity for each site according to year and species. Mean clutch size, hatchability and number of young produced to nest leaving (± standard error) for successful nests.

Site	**Year**	**Species**	**Total Eggs Laid (total nests)**	**Clutch Size (n)**	**Hatchability (n)**	**Total Young Produced (average # eggs hatched / nest)**	**# Young Leaving/Nest (n)**
BEAR	2005	AMAV	715 (311)	3.92 ± 0.67 (143)	0.96 ± 0.10 (143)	536 (1.7)	3.75 ± 0.72 (143)
		BNST	94 (29)	3.9 ± 0.57 (10)	0.98 ± 0.06 (10)	38 (1.3)	3.8 ± 0.42 (10)
	2006	AMAV	924 (302)	3.92 ± 0.52 (171)	0.94 ± 0.15 (151)	596 (1.97)	3.68 ± (162)
		BNST	84 (23)	4 ± 0 (18)	0.91 ± 0.15 (18)	65 (2.8)	3.61 ± (18)
FARM	2005	AMAV	1681 (481)	3.86 ± 0.51 (247)	0.96 ± 0.13 (247)	914 (1.9)	3.75 ± 0.57 (247)
		BNST	769 (411)	3.87 ± 0.48 (201)	0.97 ± 0.11 (201)	737 (1.79)	3.76 ± 0.62 (201)
	2006	AMAV	2146 (641)	3.93 ± 0.30 (413)	0.93 ± 0.15 (369)	1538 (2.4)	3.55 ± (435)
		BNST	1123 (313)	3.97 ± 0.21 (232)	0.96 ± 0.12 (221)	916 (2.9)	3.77 ± (243)
ISSR	2006	AMAV	507 (158)	3.9 ± .037 (42)	0.98 ± 0.08 (29)	122 (0.77)	3.59 ± (34)
		BNST	22 (8)	4 ± 0 (3)	-	4 (0.5)	4 ± 0 (1)
SHORE	2005	AMAV	18 (6)	4.0 ± 0.0 (3)	-		-
		BNST	-	-	-		-
	2006	AMAV	295 (106)	3.88 ± 0.33 (25)	0.89 ± 0.16 (14)	60 (0.57)	3.53 ± (17)
		BNST	20 (7)	4 ± 0 (4)	0.94 ± 0.13 (4)	15 (2.14)	3.75 ± (4)
SL CANAL	2005	AMAV	36 (11)	3.6 ± 0.70 (10)	1 ± 0.0 (5)	16 (1.45)	3.2 ± 0.84 (5)
		BNST	61 (16)	3.81 ± 0.54 (16)	0.98 ± 0.07 (13)	47 (2.9)	3.62 ± 0.65 (13)
	2006	AMAV	61 (19)	3.71 ± 0.76 (7)	1 ± 0 (8)	31 (1.63)	3.88 ± (8)
		BNST	-	-	-	-	-

Table 3.3.2. Mean aggregate % volume of food items recovered from the digestive tracts of American Avocets and Black-necked Stilts. See Appendix C for more details.

Taxa	AMAV N = 31	BNST N = 41
	Mean Aggregate % Volume	Mean Aggregate % Volume
Gastropoda	0.4	1.6
Odonata	0.2	5
Hemiptera		
Corixidae	23.2	30
Coleoptera		
Carabidae	3	0.6
Dytiscidae	0	2
Hydrophilidae	4.7	7.5
Coleoptera Parts	3	10.5
Trichoptera		
Limnephilidae	0.1	0
Diptera		
Culicidae	0.8	0.5
Ceratopogonidae	0	0.2
Chironomidae	33.7	17.2
Stratiomyidae	0	0.01
Syrphidae	0	3.6
Ephydridae	6.1	5.6
Muscidae	1.4	3.3
Misc. Diptera	0	2.6
Hymenoptera		
Braconidae	0.9	0.01
Seeds	15.2	4.2
Unidentifiable Parts	7	5.2



AMAV BNST

Figure 3.3.1. Volumetric proportion of food items recovered from the digestive tracts of American avocets and Black-necked stilts collected from Bear River Migratory Bird Refuge.

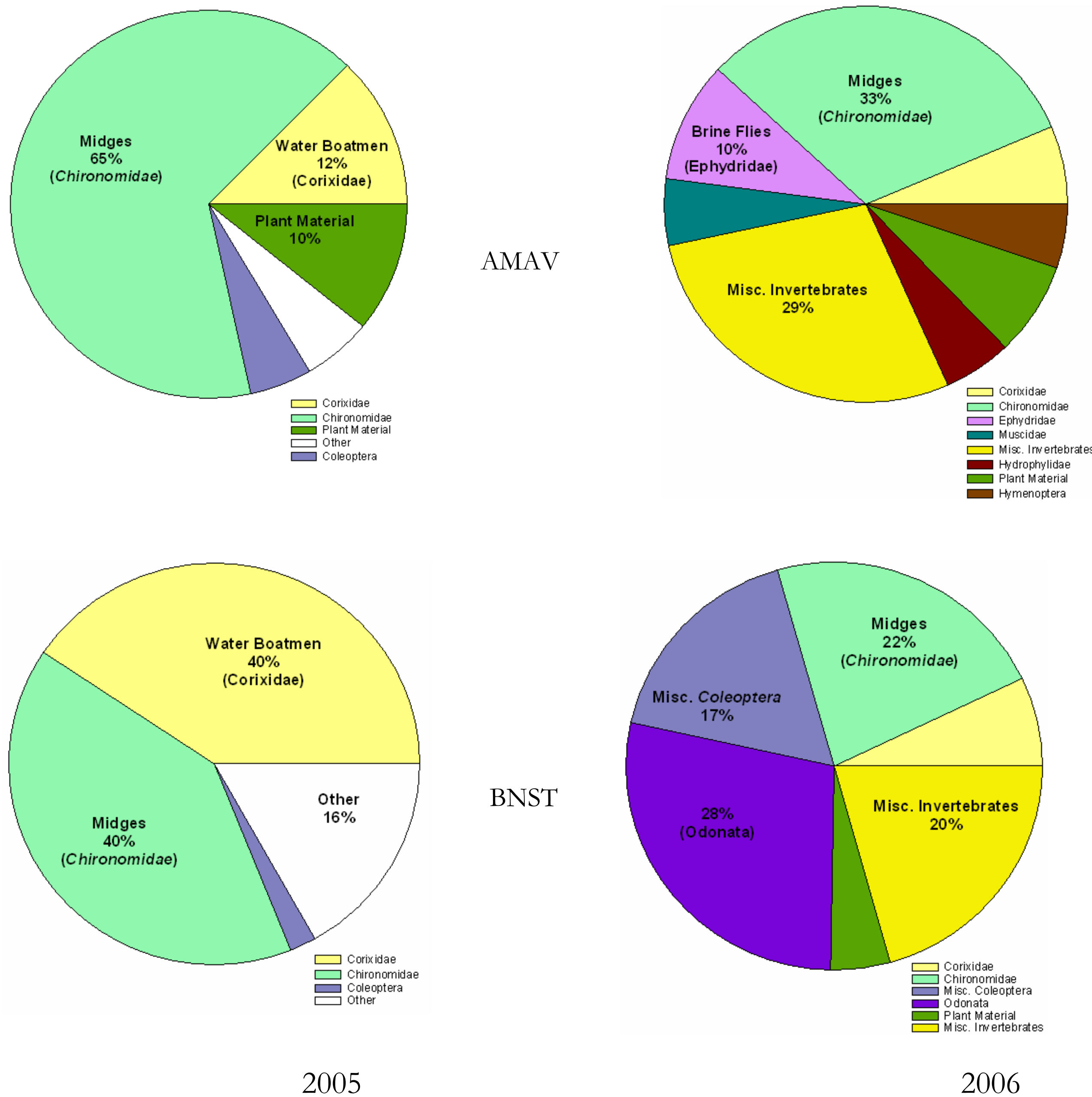
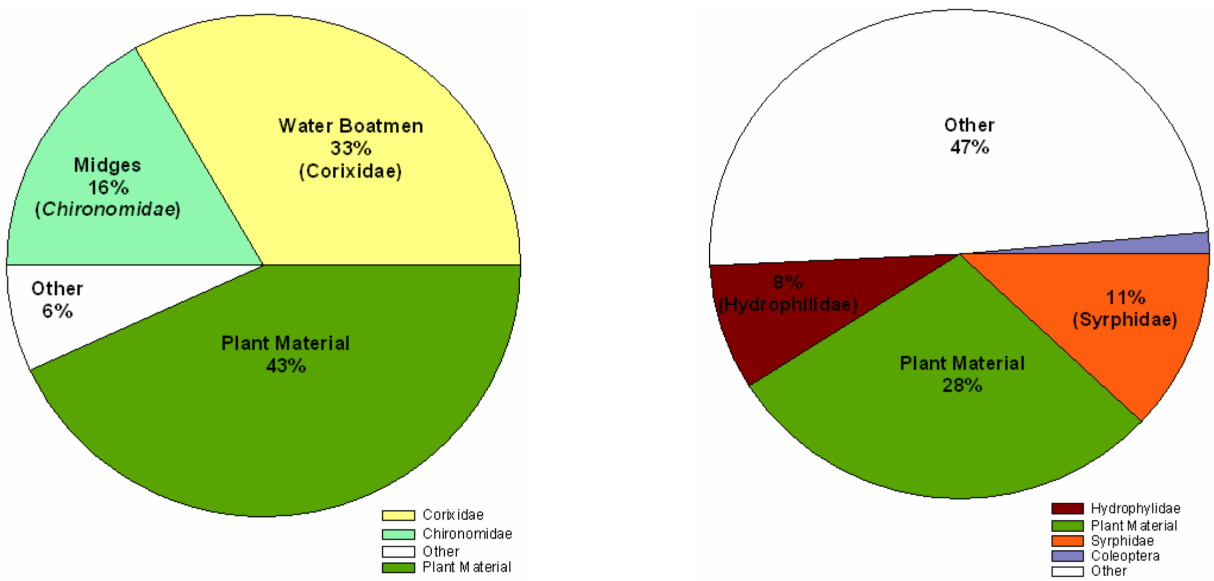


Figure 3.3.2. Volumetric proportion of food items recovered from the digestive tracts of American avocets and Black-necked stilts collected from Farmington Bay Waterfowl Management Area near the Turpin Dike.



AMAV BNST

Figure 3.3.3. Volumetric proportion of food items recovered from the digestive tracts of American avocets and Black-necked stilts near the Central Davis Sewer Discharge.

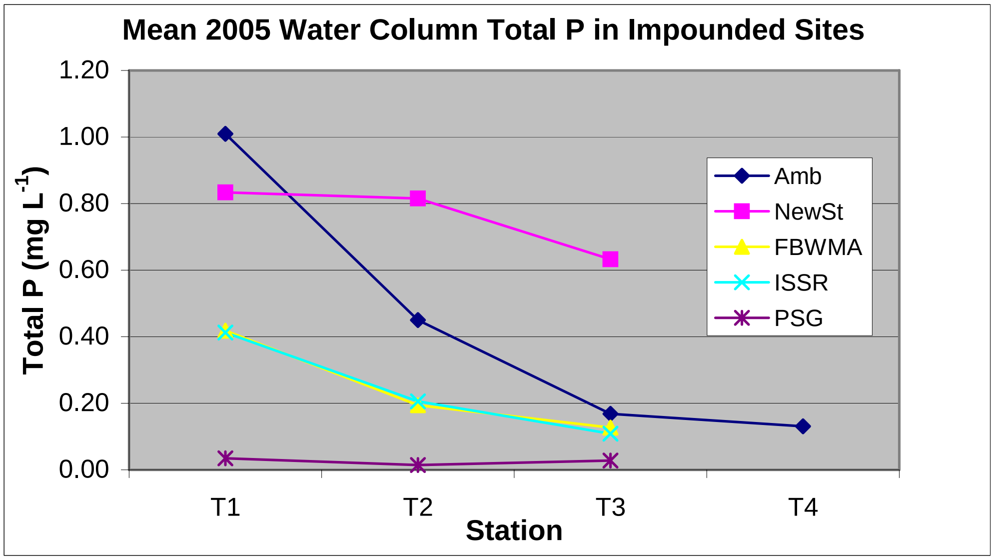
3.4 Water Column and Sediment Phosphorus Dynamics

One of the common paradigms of wetland function is the processing and reduction of nutrients from the water column (see Kadlec and Knight 1996). Indeed, one of the central tenets of our study design was to track the expected reductions in nutrients as water flows across the mudflats as sheetflow or through the successive impoundments built by the duck clubs and wildlife management areas. In reality, however, our observations did not support this hypothesis. With regard to N, except for the Newstate Duck Club ponds and the first pond of Ambassador Duck Club, water column nitrate-nitrite was nearly always below the detection limit (0.05 mg L^{-1}) at the impounded sites.

With regard to water column P, there was only slight reduction in concentrations throughout the successive ponds at the impounded sites (Figure 3.4.1). The only exception occurred among the four study ponds in the Ambassador Duck Club. In these ponds total water column P fell from a mean of greater than 1 mg L^{-1} at T-1 to about 0.1 mg L^{-1} at T-4. The primary reason for this substantial P reduction at Ambassador versus other target sites is a long water retention time in the Ambassador lower ponds. Consequently, estimated P loading rates in Ambassador ranged from about 10 g m^{-2} at T-1 to about 0.5 g m^{-2} at T-4 (Rino Decataldo, unpublished data). As a result, water and sediment concentrations of the Ambassador ponds declined substantially with each successive pond, as there was more time for assimilation. Notably, water and sediment P concentrations in Ambassador T-4 were the lowest of any sample site in Farmington Bay (Figures 3.4.2) and the sediment P concentrations in Ambassador T-4 was actually slightly less than those in the reference ponds of PSG. In contrast, estimated loading rates for Newstate Duck Club and FB WMA remained between 6 and 10 g m^{-2} at all ponds. Consequently, considerable P remained in the water column and passed from pond to pond in these other target systems.

In addition to a much shorter water retention time, the apparent lack of nutrient attenuation in the water column at most targeted sites is also attributed to saturation of binding sites in the sediments. Sediments collected from our sampling stations contained from 280 to 585 mg kg^{-1} total P. Most notably, biologically available (soluble) P ranged from 10 to 80 mg kg^{-1} in the sediments (data not shown). This readily available supply of P indicates that P concentrations between water and sediments are at equilibrium and explains why water column concentrations remained elevated (0.4 to 4 mg L^{-1}) throughout the targeted impoundments. These characteristic high P concentrations are likely responsible for the impacts described in Section 3.2 above.

At sheetflow sites, water column P in the target (POTW effluent) sites did not experience reductions in P concentrations as it progressed across the mudflats (except for a moderate reduction in the wetlands below the CDSD, Figure 3.4.1). Again, it is likely that sediment-binding sites for phosphorus and the assimilative capacity of wetland vegetation are saturated and further nutrient reduction is minimal. The high values for biologically available P and the high release rates of sediment samples suffused by various water sources (Appendix F; See Section 3.5 below) demonstrate that there is free exchange of P between sediment and the overlying water.



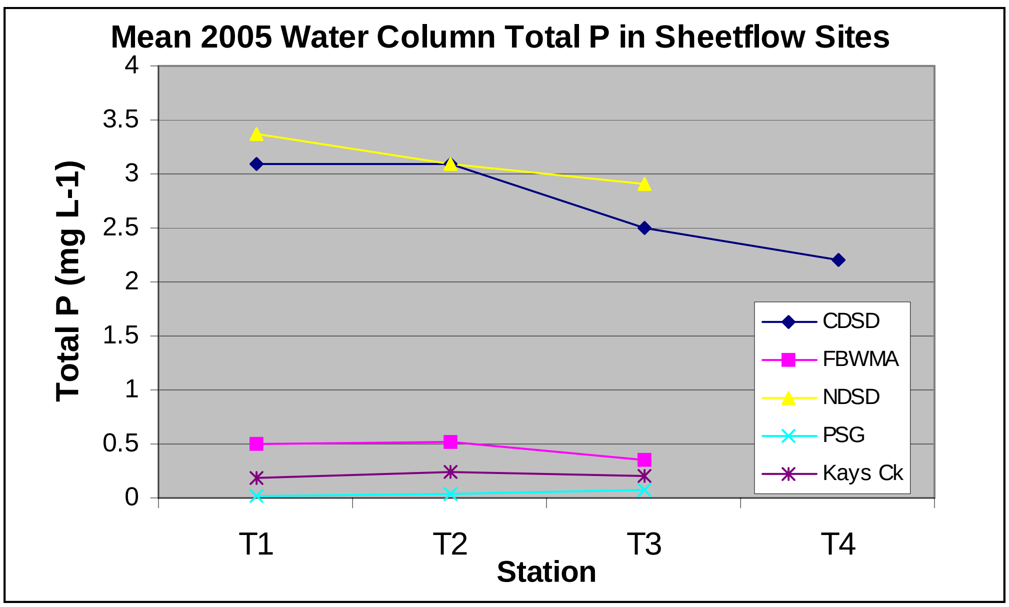
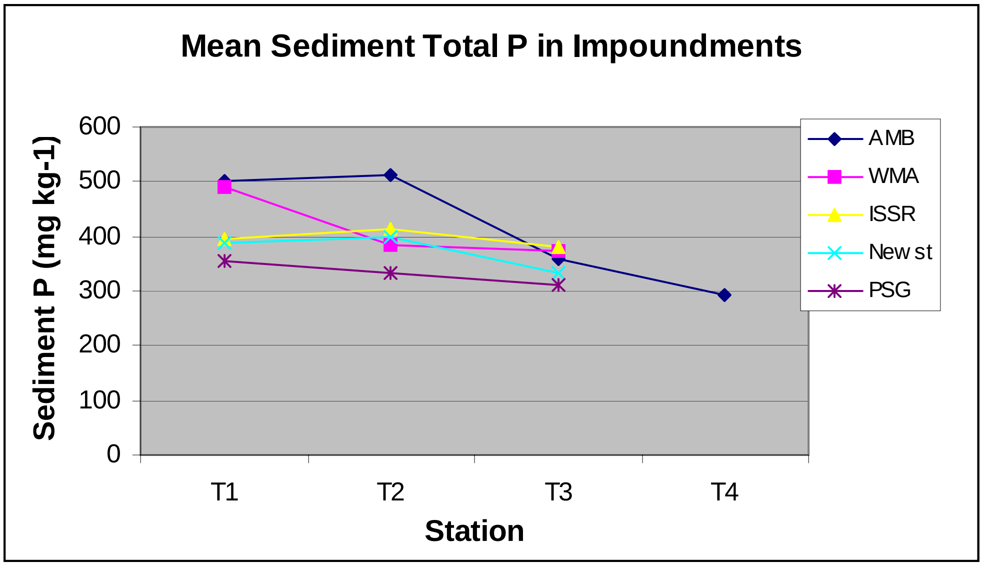


Figure 3.4.1. Phosphorus concentrations in water samples collected along impounded (upper) and sheetflow (lower) transects. (Amb – Ambassador Duck Club, Newst = Newstate Duck Club, CDSD =Central Davis Sewer District, NDSD =North Davis Sewer District, FBWMA – Farmington Bay Wildlife Management Area near Turpin Dike, PSG = Public Shooting Grounds Wildlife Management Area). Reference sites were located at Kays Creek and Public Shooting Grounds.



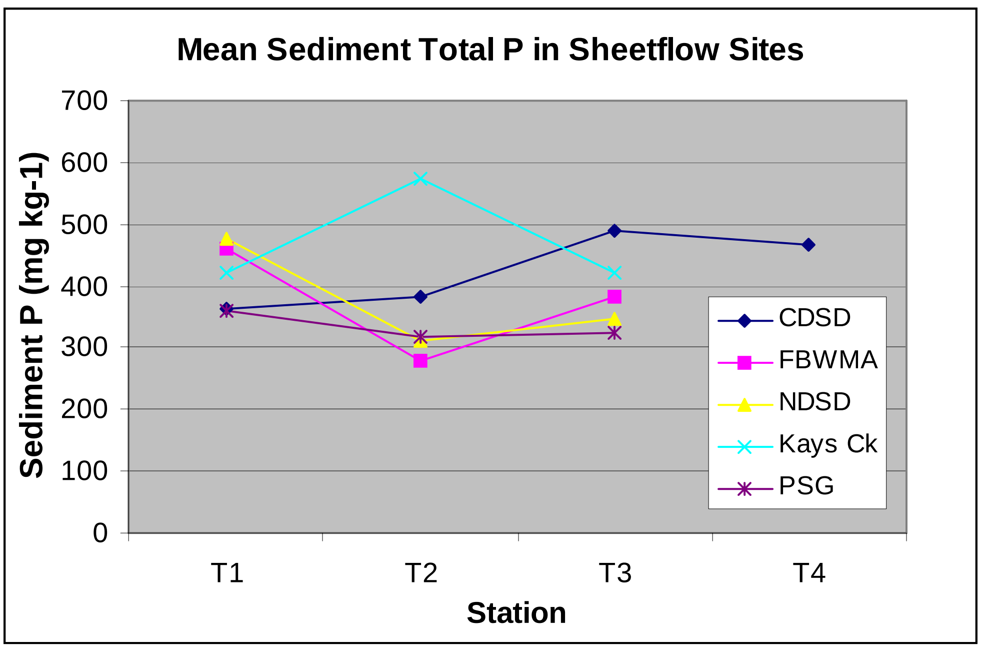


Figure 3.4.2. Sediment P concentrations at our impounded (upper) and sheetflow (lower) study sites. Reference sites were located at Kays Creek and Public Shooting Grounds Wildlife Management Area.

Perhaps the greatest insight into water/sediment nutrient relationships and wetland nutrient assimilation has been gained through the investigation of treatment wetlands. These results and subsequent treatment wetland design have been reviewed by Kadlec and Knight (1996), Faulkner and Richardson (1989), Richardson and Marshal (1986), Nichols (1983) and others. Suggested design of treatment wetlands includes loading rates of 0.5 to 3 mg P kg^{-1} yr^{-1}. Successful retention of P is dynamic and is primarily related to the amount of sediment Fe and Al and secondarily to Ca. Depending upon the sediment concentrations of these metals and loading rate, retention capacity is usually reached within the first 5 to 8 years. In other words, with a loading rate of 2 to 4 g P m^{-2} y^{-1} 90 to 95 % retention can be accomplished for the first few years. After that time retention is negligible and is primarily related to burial of organic debris (Kadlec and Hammer 1983). As expected, however, this burial can be enhanced with elevated concentrations of Fe, Al and Ca in the water as these metals are known to form organo-metal-P complexes (e.g. $R(COO)_3Al\ H_2PO_4$ where R represents any carboxylated compound, although humic and fulvic acids are the most commonly mentioned (Bloom 1981).

These processes, however, may be quite variable depending upon vegetation type and nutrient concentrations in the water. In emergent wetland systems (such as our sheetflow sites), the primary source of nutrients is sediments (see discussion in Section 3.2). Klopatek (1978) determined a nutrient budget for the emergent *Scheonplectus fluviatilis* and found 3.8 g m^{-2} yr^{-1} were translocated from the wetland soil to the plant shoots. At the end of the growing season about 12% of this P was transferred to the roots and stored over winter. Fifty eight percent of the P was leached into the water column during senescence and the rest remained associated with the dead plant material. Prentki (1978) found similar results in a Wisconsin cattail marsh. This process actually promotes a net annual movement of nutrients out of the sediment and into the water column. Wetlands dominated by SAV may behave similarly. For example, the above ground biomass is nearly completely decomposed within the water column and thereby releases similarly large amounts of nutrients back into the water column (Nichols and Keeney, 1973, Barko and Smart 1980, Brenner et al. 2006).

Notably, samples from the last pond of the Ambassador Duck Club complex (T-4) had the lowest water and sediment P, including those collected from the pelagic zone of Farmington Bay (Figures 3.4.1, 3.4.2 and Appendix F). The reduction of water column P is attributed to the much greater water retention time in the Ambassador ponds, allowing more efficient sorption and sedimentation of P. However, this doesn't explain why sediment concentrations are so low. Rather, annual macrophyte production, including obtaining the majority of P from sediments, followed by winter senescence and leaching to the water column and subsequent pond flushing at the end of the hunting season, would provide an annual net loss of sediment P. This would explain the successively lower sediment P that was measured in Ambassador T-3 and T-4.

In contrast, sediment samples from the CDSD transects had the highest P and N concentrations – suggesting that removal of P by macrophyte growth and senescence is readily replaced (or surpassed) by sorption and sedimentation of P from the effluent itself. Indeed, there is elevated sediment P along the eastern fringe of Farmington Bay that is either associated with POTW discharges and/or gradual burial of detritus. Stable isotope analysis such as described by Brenner (2006) may elucidate the whether the source of nutrients is from (internal) wetland sources, POTWs or from other tributary sources.

3.5 Water-Sediment Interactions

A series of experiments were conducted to assess the ability of Farmington Bay sediments to release P back into the water column. This ability has serious implications for present or future nutrient management decisions regarding Farmington Bay and its wetlands. Toward this goal CDSD, in conjunction with USGS, collected several dozen core samples from throughout Farmington Bay (Houston et al. 2006.; Appendix F). Sampling sites were selected in both littoral/wetland environments and pelagic sites.

Cesium dating of sediment cores indicates that approximately 0.4 cm of sediment is added annually to Farmington Bay. Further, P analysis in sediment cores indicates that high loading to Farmington Bay has occurred since before modern settlement (>150 years). Throughout the many core samples, P concentrations ranged from 400 to 1200 mg kg^{-1} sediment (data not shown). Highest concentrations occurred along the wetland fringe of the eastern shoreline and decreased with distance toward the west. Similarly, samples collected near the wetlands contained elevated P in the top 3-5 cm. Otherwise, phosphorus concentrations were quite uniform throughout the core sample.

Several tests were conducted to determine P transfer between sediment and water using aerobic and anaerobic sediments and four sources of fresh water: deionized, Kays Creek, CDSD effluent, and NDSD effluent. This was performed by placing a small amount of sediment (approximately 1 g wet weight), into 5 ml centrifuge tubes. Approximately 4 ml of water was then placed in the tubes followed by shaking for 1 min. The samples were then centrifuged for 30 sec and analyzed using a HACH DR-4000 spectrophotometer (Handbook, Method 8048). Enough replicates were prepared to provide P measurements at several time intervals, ranging from 5 min to 24 hr.

In one set of experiments, an aerobic sediment sample collected from the area between the CDSD discharge and the Farmington Bay WMA Unit one discharge and was suffused with either deionized water or with 100% Central Davis Sewer District effluent (Figure 3.5.2). This sediment sample sequestered a significant amount of phosphorus from the effluent water – until the final water concentration reached about 2 mg L^{-1}. However, when suffused with deionized water, this sediment gave up significant amounts of phosphorus – until the final water concentration reached about 1 mg L^{-1}. Interestingly, the average surface water P concentration at our sampling stations along the horizontal transect that follows the Central Davis effluent remained at about 2 mg L^{-1}.

In another experiment, an aerobic sediment sample collected adjacent to Antelope Island was suffused with either Kays Creek water or North Davis Sewer District effluent (Fig.) 3.5.3). Sediment P at this site was relatively very low (circa 300 mg kg^{-1}). Total P in Kays Creek water (background P concentrations ranging from 0.1 to 0.4 mg L^{-1}) remained stable at about 0.2 mg L^{-1} while the P concentration in NDSD water fell by 1.7 mg L^{-1} (from approximately 3.7 mg L^{-1} to 2 mg L^{-1}) to the aerobic sediments.

Anaerobic sediments reacted differently. Both Kays Creek and effluent water gained phosphorus from anaerobic sediments (Fig. 3.5.4). After six hours, both water sources contained between 5 and 6 mg L^{-1}.

Although these data are preliminary, they suggest that internal loading from anaerobic sediments may be substantial (although the assimilative capacity of vegetation growing in anaerobic sediments would counteract the internal loading somewhat). It should be noted that equilibrium is reached experimentally within a relatively short time. These studies support the explanation of why there was very little attenuation along the various longitudinal transects in our sheetflow study sites, i.e. saturation of sediment binding sites has been reached or surpassed – allowing re-entry of pore water or loosely-bound P into the water column.

Observations of P being released from either anaerobic or aerobic sediments strongly suggests that there is substantial amounts of pore-water P and/or P is loosely adsorbed to clay or silt particles or to organic material in the sediment rather than the more commonly described dissolution of Fe(OOH)≈P with reducing condition (Mortimer 1941, 1942, Van Lier, et al. 1983).

Some of this P release or sequestering in organic-enriched lakes and wetlands can be mediated by the microbial communities that utilize various forms of organic carbon (by either mineralization or bacterial growth). (e.g. Kelton et al. 2004). Organic carbon at our sampling sites was variable but quite high (1 to 5%). Therefore, it is likely that part of the sediment/water equilibrium is microbially mediated. However, there is little information as to the permanency of this relationship and particularly on a long-term basis.

Because Farmington Bay wetland and pelagic sediments have similarly high P and organic carbon, microbial processes and the more-labile organo-metal-P complexing may reflect the dynamic movement of P into or out of Farmington Bay sediments and play a major role in the equilibrium process.

Finally, the work of Brenner et al. 2006 may provide further insight into the complex sediment/water equilibrium processes. They reported that recent nutrient enrichment in shallow mesotrophic (mean TSI = 47) Lake Panasoffkee, Florida resulted in an increased presence of SAV. In view of the previous discussion, this would suggest that sedimentation of P might decline and perhaps even cause a net removal of P from sediments after the growth and senescence cycle. Yet, sediment P concentrations actually increased. Further, they linked recent carbon sedimentation to an increase in algal decomposition and sedimentation. This presents an apparent contradiction to the paradigm that most macrophyte dominated shallow lakes and wetlands support a low phytoplankton biomass. However, they proposed an interesting hypothesis that explains this contradiction by suggesting that the source of algae is the substantial epiphytic populations associated with the SAV. In turn, considerable photosynthesis of the epiphytic community resulted in localized elevation in pH and precipitation of nucleated calcite crystals or Ca-organic P complexes. They hypothesized that the primary sedimentation process occurs when encrusted carbonate sloughs off the leaves of higher plants, thereby delivering epiphytes along with organic- and carbonate-bound P to the sediments. Evidence supporting this process was obtained from C:N ratio data and the $\delta^{15}N$ isotopic signature which indicated that the organic C and N in the sediment were derived from algal sources rather than macrophytic tissue. In support of this hypothesis, in a

CDSD Sediment Water Interaction

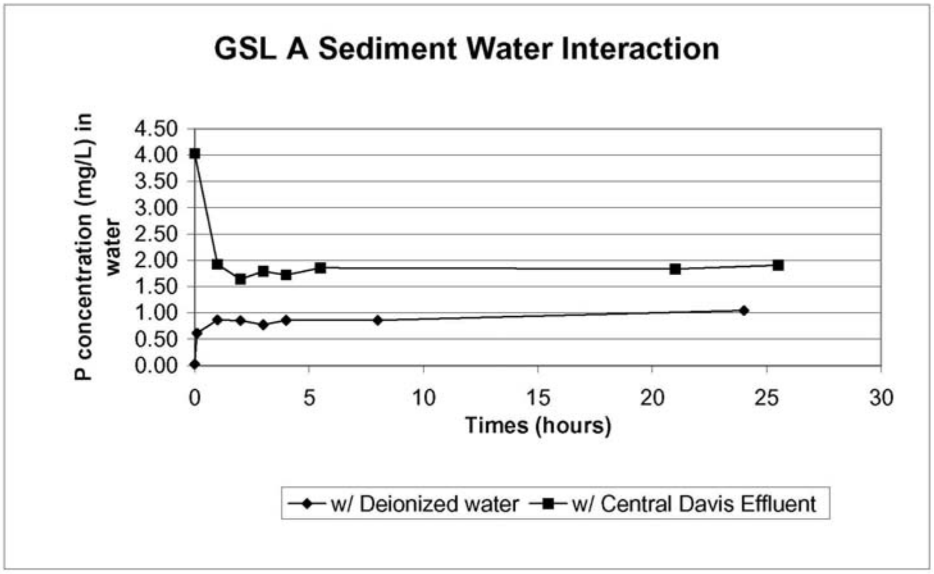
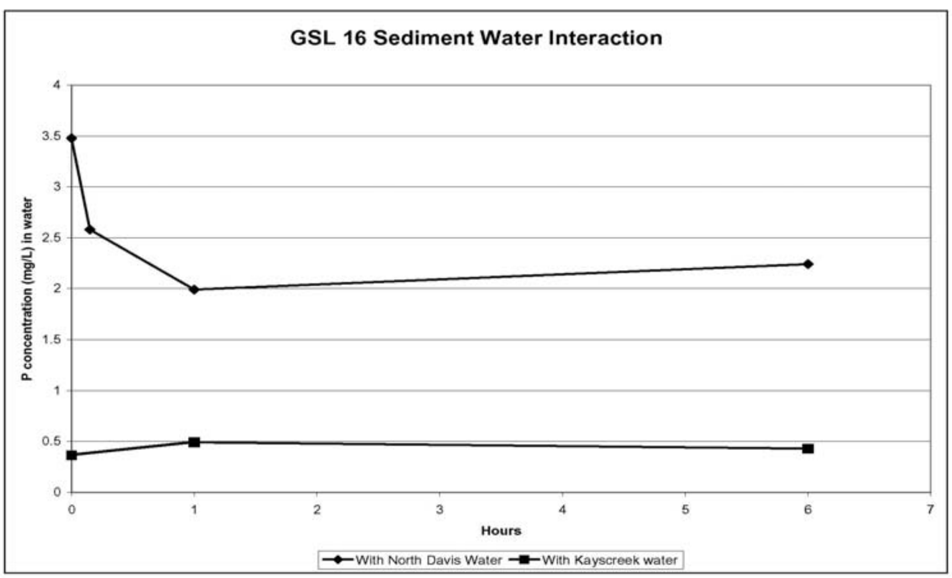


Figure 3.5.2. Accumulation of water column P concentrations after suffusing an aerobic Farmington Bay sediment sample with deionized water or Central Davis Sewer District effluent. The sediment sample was collected in the emergent wetland area approximately 2.5 km south of the Central Davis Sewer outfall. P concentrations and Time 0 = the initial concentrations before the water was applied.



ata-1

Figure 3.5.3. Accumulation of water column P concentrations after suffusing an aerobic sediment sample with either Kays Creek water or North Davis Sewer District effluent. The sediment sample was collected from a site near Antelope Island. P concentrations at Time 0 = the initial concentrations before the water was applied.

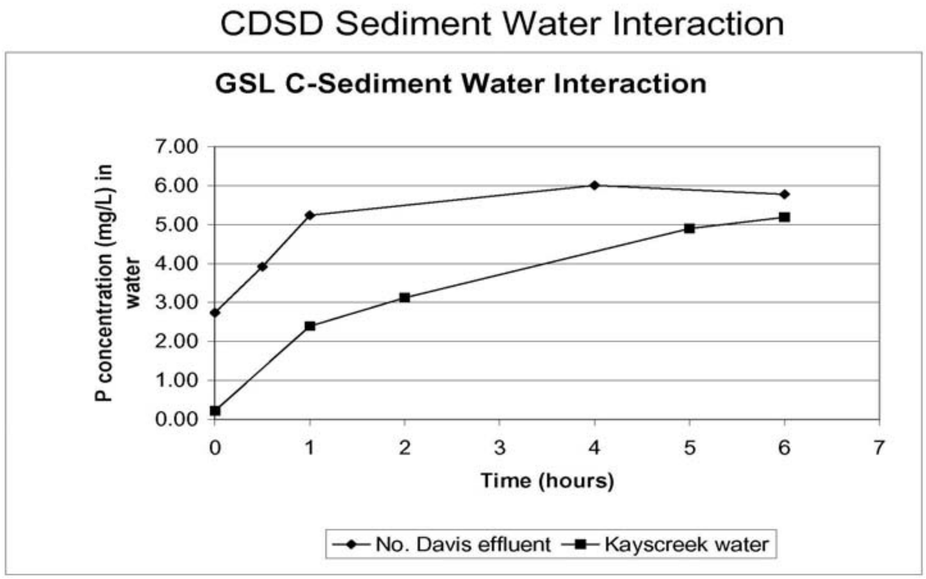


Figure 3.5.4. Accumulation of water column P concentration after suffusing an anaerobic sediment sample with either Kays Creek or North Davis effluent water. The sediment sample was collected from near the North Davis Sewer District discharge. P concentrations at Time 0 = the initial concentrations before the water was applied.

sample of 12 cores from five Florida lakes where similar C:N measurements and isotopic analysis was performed, total P was 2.6-fold greater in sediments derived from phytoplankton compared with sediments formed by macrophytes (Kenney et al. 2002). Because of the high calcium concentration in Farmington Bay tributary water, this may be an attractive hypothesis for nutrient management of the impounded target sites of Farmington Bay. For example, in Lake Panasoffkee, Florida, P concentrations generally averaged 0.1-2 mg L^{-1} (CM2MHill 1995). Our impounded sites encompass these values (i.e. total P in PSG impounded sites ~0.03 mg L^{-1} and target sites in Ambassador ranged from 1 mg L^{-1} down to 0.15 mg L^{-1}). Therefore careful sampling of water and sediment quality, including C:N:P ratios and stable isotopes of epiphytic algal communities could provide support for this hypothesis.

3.6 Conclusions

One of the major metrics suggested in EPA's "*Methods for Evaluating Wetland Condition*" modules is changes in species composition to invasive/exotic species and a reduction in species richness. Indeed, for the sheetflow sites, many measurements of the plant community were inversely related to water and soil pH. These included cattails and Phragmites % cover, and *Scirpus americanus* and *Distichilis spicata* stem height. However, diversity was actually higher in the fresher, more nutrient-rich sites. Moreover, this diversity was a result of non-native or aggressive invasive species. Those sites that were more proximal to the discharge points were dominated by native but aggressive cattails and Phragmites. The more-distal sites were dominated by native non-aggressive alkali bulrush (*Scirpus americanus)* and secondarily by pickleweed (*Salicornia* spp.). These two taxa were dispersed by seeds and, along with their relatively high tolerance to salinity, explains why these taxa were the first to colonize and rapidly expand across the mudflats as the lake receded and salts were successively leached from the sediments. On the other hand, stands of cattails and Phragmites expand primarily by rhizomes and are known to eventually shade out the shorter bulrush species. Phragmites and cattails are expanding across the mudflats and almost exclusively follow the freshwater flows. These taxa will likely continue to expand their dominance as sediment salts continue to be flushed by fresh water. These contrasting results are uniquely dependent upon the duration and intensity of freshwater leaching and ultimately leads to the possibility that, if the lake were to remain at relatively low levels, the mudflats will eventually become dominated by these two invasive and generally less desirable species.

Macroinvertebrate taxa that are tolerant of organic and nutrient enrichment were predictably dominant in the targeted sites. These include chironomids and corixids. Other taxa exhibited sensitivity to the nutrient gradient, including mayflies and odonates. These taxa are candidates for inclusion in the list metrics that will be developed for the assessment and the standard-setting process.

Although chironomids and corixids were generally dominant among the targeted sites, they were also the most common food items eaten by shorebirds. In addition to the observed high nesting and hatching success, the predominance of these taxa as food items suggests that shorebird populations are in a healthy condition.

For the impounded sites, the submergent Stuckenia (sego pond week) was the predominant taxa among both the targeted and reference sites. These ponds are intensively managed for this species because it is the most desirable forage species for waterfowl. Therefore, the early senescence of Stuckenia is of particular interest and concern because it provides a direct link to beneficial use support for waterfowl. This may be one of the most important measures for standard setting as well as an easily obtainable metric for biological monitoring. Therefore, it is imperative that plant density and persistence and associated measurements of surface mats and epiphytes be performed in order to confirm this observation and elucidate these complex relationships.

The considerable exchange between sediment and water in wetland and pelagic environments has huge implications for the potential management alternatives and decisions that will be made in regard to point and nonpoint source limits. Further, because of the potentially enormous financial requirements that would be necessary to reduce P inputs into Farmington Bay, the recycling of sediment P in the wetlands, both in aerobic and anaerobic sediments, warrants considerably more study to ensure that reduction of external loading is cost-effective and will improve water quality. For example, such decisions were faced by managers of a hypereutrophic shallow lake system in the UK, with similarly high sediment P (circa 1000 mg kg^{-1}, Phillips et al. 1994). Several million dollars were spent in reducing P inputs from point sources. Yet, in 1992, twelve years since achieving a 90% reduction in external phosphorus load, there was little reduction in water column P or primary production. They determined that peak internal phosphorus sources in this shallow lake system was still as high as 130 mg P m^{-2} d^{-1}, compared with an external load of only 12 mg P m^{-2} d^{-1}. They attributed the internal P source to organically-bound P, which formed at least 50% of the total sediment P, and likely involves the organo-metal-P complexes discussed above.

Several measures described in this report have demonstrated sensitivity to nutrient or turbidity gradients and are candidates for inclusion into a multimetric index of biological integrity for wetlands assessment. Moreover, we have made particular effort to select parameters that have a direct relationship to the beneficial uses identified for the wetlands. In addition to the data gap recommendations identified in Section 3.1.3, these measures will contribute to data set that is essential for establishing appropriate site specific nutrient criteria for these wetlands. A summary of these measures include:

1. Macroinvertebrate species composition and density (during nesting season and fall migration season).
2. Percent of Ephemeroptera
3. Percent of Chironomidae
4. Percent Odonates or clingers
5. Percent exotic and/or invasive plants
6. Submerged aquatic vegetation above ground biomass
7. SAV percent coverage
8. C:N:P ratios in phytoplankton and macrophytes
9. SAV leaf Chlorophyll a / macrophyte fluorescence
10. turbidity/ light penetration
11. Presence/composition of floating vegetation
12. Presence/composition of SAV epiphytes
13. Summer mean diel DO
14. Diel minimum DO
15. Water column and sediment H_2S measurements

Finally, Reports by Rushforth (Appendix D) and Wurtzbaugh (Appendix E) are also appended to this report to display the additional research that has been performed as part of this grant and program. However, detailed analysis and interpretation, such as presented here, is not included in this report. Rather, additional data collection, analysis and reporting will be provided by the end of 2007. As a result, some algal measures may be added to the list of potential metrics.

4.0 Literature Cited

Anscombe, F.J., 1948. The transformation of Poisson, binomial, and negative binomial data. Biometrika 35:246-254.

Barko, J.W. and R.M. Smart. 1980. mobilization of sediment phosphorus by submerged freshwater macrophytes. Freshwater Biol. 10:229

Bloom, P.R. 1981. Phosphorus adsorption by an aluminum-peat complex. Soil Sci. Soc. Am. Proc. 45:267-272.

Brenner, M., D.A. Hodell, B.W. Leyden, J.H. Curtis, W.F. Kenney, B. Gu and J.M. Newman. 2006. Mechanisms for organic matter and phosphorus burial in sediments of a shallow, subtropical, macrophyte-dominated lake. Journal of Paleolimnology 35: 129–148

Brix, H. and JE Lyngby. 1983. The distribution of some metallic elements in eelgrass (Zostera marina L.) and sediment in the Limfjord, Denmark. Estuaries and Coastal Shelf Science 16:455-467.

Burrel, D.C. and J.R. Schubel. 1977. Seagrass ecosystem oceanography. In: C.P. McRoy and C. Helfferich (eds.) Seagrass ecosystems: A scientific perspective. Marcel Dekker, Inc. New York. Pp. 195-232.

CH2M Hill, Inc. 1995. Lake Panasoffkee Water and Nutrient Budget Study. Final Report to the Southwest Florida Water Management District. Tampa.

Canfield, D.E. and M.V. Hoyer. 1988. Influence of nutrient enrichment and light availability on the abundance of aquatic macrophytes in Florida streams. Can. J. Fish. And Aquat. Sci. 45:1467-1472.

Carignan, R. and J. Kalff . 1980. Phosphorus Sources for Aquatic Weeds: Water or Sediments? Science. 29 (207):987–989.

Carr, G.M. and P.A. Chambers. 1998. Macrophyte growth and sediment phosphorus and nitrogen in a Canadian prairie river. Freshwater Biology. 39:525-536.

Duong, T.P. and J.M. Tiedje. 1985. Nitrogen fixation by naturally occurring duckweed-cyanobacterial associations. Can. J. Microbiol. 31(4): 327-330.

Finke, L.R. and H. W. Seeley.1978. Nitrogen fixation (acetylene reduction) by epiphytes of freshwater macrophytes. Applied and Environmental Microbiology. 36 (1): 129-138.

Good. R. E., D. F. Whigham and R. L. Simpson, (eds.), Academic Press, New York.

Güsewell, S. and W. Koerselman. 2002. Variation in nitrogen and phosphorus concentrations of wetland plants In: Perspectives in Plant Ecology, Evolution and Systematics, Volume 5, Number 1, Urban & Fischer, Germany; pp. 37-61(25).

Hoven, H.M., H.E. Gaudette and F. T. Short. 1999. Isotope ratios of 206Pb/207Pb in eelgrass, *Zostera marina*, indicate sources of Pb in an estuary. Marine Environmental Resources 48: 377 - 387.

Kadlec, R.H. and R.L. Knight, 1996. Treatment Wetlands. New York: CRC Press.

Kemp, W.M., R.R. Twilley, W.R. Boynton, and J.C. Means, 1983. The decline of submersed vascular plants in upper Chesapeake Bay: Summary of results concerning possible causes. Marine Technology Society Journal 17:78-79.

Kelton, N., P. Chow-Fraser and I. Jordan. 2004. Relationship between sediment phosphorus release rates and characteristics of the benthic microbial community in a hypereutrophic marsh. Aquatic Ecos. Health and Managmnt. 7(1): 31-41.

Kenney, W.F., M.N. Waters, C.L. Schelske and M. Brenner. 2002. Sediment records of phosphorus-driven shifts to phytoplankton dominance in shallow Florida lakes. J. Paleolimnol. 27:367–377.

Klopatek, J.M. 1978. Nutrient dynamics of freshwater riverine marshes and the role of emergent macrophytes. IN: Freshwater wetlands. Ecological processes and management potential. R.E. Good et al. (eds.) Academic Press. New York, NY

Madden, C.J., and W.M. Kemp. 1996. Ecosystem Model of an Estuarine Submersed Plant Community: Calibration and Simulation of Eutrophication Responses. Estuaries Vol. 19 No. 2B:457-474.

Madsen J.D. and M.S. Adams. 1988. The nutrient dynamics of a submersed macrophyte community in a stream ecosystem dominated by *Potamogeton pectinatus.* J. Freshwat. Ecol., 4:541-550.

Morris, K., K.A. Harrison, P.C.E. Bailey and P. Boon. 2004. Domain shifts in the aquatic vegetation of shallow urban lakes: the relative roles of low light and anoxia in the catastrophic loss of the submerged angiosperm *Vallisneria Americana.* Marine and Freshwater Research 55(8): 749–758

Mortimer, C.H. (1941). The exchange of dissolved substances between mud and water in lakes. I. *J. Ecol.*, **29**, 280-329.

Mortimer, C.H. (1942). The exchange of dissolved substances between mud and water in lakes II. *J. Ecol*., **30**, 147-201.

Nichols, D.S. 1983. Capacity of natural wetlands to remove nutrients from wastewater. J. Water Pol. Cont. Fed. 55(3): 495-504.

Nichols, D.S. and D.R. Keeney. 1973. Nitrogen and phosphorus release from decaying water milfoil. Hydrobiologia. 42:509

Orthe, R.J., and K.A. Moore. 1983. Chesapeake Bay: An unprecedented decline in submerged aquatic vegetation. Science: 222:51-53.

Peltier, W.H. and E.B. Welch. 1969. Factors affecting growth of rooted aquatics in a river. Weed Science. 17:412-416.

Portielje R, Roijackers RMM. 1995. Primary succession of aquatic macrophytes in experimental ditches in relation to nutrient input. Aquat. Bot. 50:127-140.

Phillips, G., R. Jackson, C. Bennett & A. Chilvers. 1994. The importance of sediment phosphorus release in the restoration of very shallow lakes (The Norfolk Broads, England) and implications for biomanipulation *Hydrobiologia* 275(276): 445-456.

Powell, G.V.N., W.J. Kenworthy and J.W. Fourqurean. 1989. Experimental evidence for nutrient limitation of seagrass growth in a tropical estuary with restricted circulation. Bulletin of Marine Science 44:324-340.

Prentki, R.T., T.D. Gustafson and M.S. Adams. 1978. Nutrient Movements in Lakeshore Marshes. p 169-194 In: Freshwater Wetlands: Ecological Processes and Management Potential.

Redfield, A.C., 1934. On the proportions of organic derivations in seawater and their relation to the composition of plankton. In: James Johnson Memorial Volume. (ed. R.J. Daniel). University Press of Liverpool, pp. 177-192.

Staver, L.W., K.W. Staver, J.C. Stevenson. 1996. Nutrient inputs to the Choptank River Estuary: Implications for watershed Management. Estuaries Vol. 19 No. 2B:342-358.

Short, F.T. and D.M. Burdick. 1996. Quantifying eelgrass habitat loss in relation to housing development and nitrogen loadings in Waquoit Bay, Massachusetts. Estuaries 19:730-739.

Short, F.T and C.A. Short. 1984. The seagrass filter: purification of estuarine and coastal water. In: VS Kennedy (ed.), The Estuary as a Filter. Acad. Press, pp. 395-413.

Stumpf, R.P., M.L. Frayer, M.J. Duako and J.C. Brock, 1999. Variations in Water Clarity and Bottom Albedo in Florida Bay from 1985 to 1997. Estuaries 22 No. 2B:431-444.

Taylor, D.I., S.W. Nixon, S.L. Granger, and B.A. Buckley. 1995. Nutrient limitation and the eutrophication of coastal lagoons. Marine Ecology Progress Series 127:235-244.

Taylor, D.I., S.W. Nixon, S.L. Granger, and B.A. Buckley. 1999. Responses of coastal lagoon plant communities to levels of nutrient enrichment: a mesocosm study. Estuaries Vol.22, No. 4:1041-1056.

Thiebaut, G. and S. Muller. 2003. Linking phosphorus pools of water sediment and macrophytes in running waters. Ann. Limnol. – Int. J. Lim. 39(4):307-316.

Tomasko, D.A, C.J. Dawes, M.O. Hall. 1996. The effects of anthropogenic nutrient enrichment on turtle grass (*Thalassia testudinum*) in Sarasota Bay, Florida. Estuaries Vol. 19, No. 2B:448-456.

Van Lier, L.J., Peters, A. Mutijin and L.R. Mur. 1982. Release of sediment-phosphorus, and the influence of algae growth. Hydrobiol. Bull. 16:191-200.

Vaithiyanathan, P. and Richardson C.J. 1999. Macrophyte species changes in the Everglades. Examination along a eutrophication gradient. J. Environ. Qual. 28:1347-1358.

Valiela, I., J. McClelland, J. Hauxwell, P.J. Behr, D. Hersh, and K. Foreman, 1997. Macroalgal blooms in shallow estuaries; Controls and ecophysiological and ecological consequences. Limnology and Oceanography 42:1105-1118.

Ward, L.G., W.M. Kemp and W.R. Boynton. 1984. The influence of waves and seagrass communities on suspended particulates in an estuarine environment. Marine Geology 59:85-103.

Ward, T.J., 1987. Temporal variation of metals in the seagrass *Posidonia australis* and its potential as a sentinel accumulator near a lead smelter. Marine Biology 95:315-321.

Wolfe, D.A., G.W. Thayer and S.M. Adams. 1976. Manganese, iron, copper and zinc in an eelgrass (*Zostera marina* L.) community. Radiology and energy resources. Proc. Fourth Ecol. Soc. Am., Spec. Publ. 1:256-270.

Zuberer, D.A., 1882. Nitrogen Fixation (Acetylene Reduction) associated with duckweed (Lemnaceae) Mats. Applied And Environmental Microbiology Vol. 43, No. 4: 823-828.

.